Urban brownfields and their potential for nature conservation: what citizen science tells us?

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Abstract

Urban areas are increasingly recognized for their potential to support biodiversity, particularly through sites such as parks, gardens, and brownfields. Despite their conservation value, brownfield consisting of former industrial sites are frequently overlooked or redeveloped, risking biodiversity loss. Citizen science offers a unique opportunity to document the conservation value of these sites, providing data where traditional scientific surveys may be limited or absent. This study aims at evaluating conservation value of brownfields in a post-industrial region. We found that users predominantly record observations in quarries and coal mine sites, while heavily industrialized areas tend to be overlooked or underrepresented. We also found that brownfields, especially former coal sites and quarries, support high densities of protected species. However, they also harbour significant numbers of invasive species. In contrast, residential and agricultural areas show lower conservation potential due to intensive human activities. Although the uneven sampling effort across brownfields limits the reliability of ecological assessments and prevents a consistent evaluation of their potential value across all areas, our results indicate that, where data are available, brownfields can play an important role in supporting biodiversity in urban areas.

Keywords

Citizen science, post-industrial sites, urban biodiversity, land use comparison

Introduction

Recent research emphasizes the potential of urban areas to support biodiversity by providing a wide range of ecological habitats, including refuges for species of conservation concern (Oke et al. 2021; Kowarik 2011). Urban environments foster habitat heterogeneity, resulting in a mosaic of natural or semi-natural spaces such as parks, gardens, and brownfields. While the role of urban green spaces in supporting biodiversity is widely acknowledged (Aronson et al. 2017), the conservation value of brownfield sites remains underexplored. This observation underscores the importance of investigating these areas to better understand their ecological significance (Loures & Vaz 2018).

The term brownfield encompasses a wide array of abandoned or underutilized urban sites. Similarly to vacant lots, they are vegetated spaces not used for residential or commercial purposes with restricted access. Whereas vacant lots tend to be abandoned over short timescales and may be periodically maintained, often keeping vegetation at ground-cover level (Rupprecht et al. 2015), brownfields differ by exhibiting less frequent maintenance, longer periods of abandonment, and minimal vegetation removal. Typically, the term brownfield refers to large former industrial areas that have been abandoned or partially rewilded (Bonthoux et al. 2014). Post-industrial sites are the legacy of past industrial activities that have ceased due to urban expansion, industrial decline, or changes in land-use policy. These areas often suffer from significant environmental degradation and are associated with negative impacts on surrounding property values, public health, and safety. A major barrier to their rehabilitation is the presence of real or perceived contamination, which complicates redevelopment and increases cleanup costs. These sites are often characterized by extensive impervious surfaces, modified topography and limited vegetation due to their previous industrial use. Despite their ecological and social potential, such spaces remain underutilized, partly due to their negative public image and the overlooked benefits they could offer to local communities. (Kim et al. 2018). In the UK, Preston et al. (2023) found that more than half of brownfield sites are vegetated, providing critical refuges for urban flora and fauna and contributing meaningfully to green infrastructure. These sites often feature unique conditions - such as minimal management, heterogeneity of ecological conditions, and dynamic process of recolonization resulting in various successional stages - which could enhance their conservation value (Kattwinkel et al. 2011; Macgregor et al. 2021; Phillips & Lindquist 2021). However, the spread of alien species may significantly alter species compositions and interactions, often resulting in the homogenization of urban biodiversity (Kowarik 2011; Lemoine 2016).

Despite their potential, the conservation value of brownfields is often neglected in urban planning due to limited ecological expertise in redevelopment processes (Cox & Rodway-Dyer 2023). Their perception as low-value spaces has accelerated their redevelopment, leading to their rapid depletion (Preston et al. 2023). This trend may result in species loss and the overlooking of opportunities for biodiversity conservation, underscoring the need to better document the conservation value of brownfields. Such evaluation could help reconcile their conservation potential for biodiversity with the pressures of urban redevelopment (Cox & Rodway-Dyer 2023). Assessing conservation value remains inherently complex, as no single metric can adequately encapsulate the richness, vulnerability, or functional integrity of an ecosystem. Conservation value can be assessed from multiple perspectives, including the rarity or vulnerability of species themselves, the degree of human impact on ecosystems, or the benefits nature provides to people. However, each of these approaches has limitations. For example, assessing anthropogenic impacts often relies on the notion of a "pristine" or reference ecosystem, a concept that is increasingly difficult to define in highly altered environments (Nobles 2025). Similarly, while frameworks focusing on nature's contributions to people have gained traction, they risk legitimizing the commodification of nature and the continued exploitation of vulnerable species (Nobles 2025). To inform conservation prioritization, widely used indicators such as species richness, species diversity, and the presence of threatened or protected species provide valuable, albeit partial, insights into ecological significance (Willis et al. 2012; Dyer et al. 2017). In urban contexts characterized by high levels of biological invasions, incorporating the occurrence and abundance of invasive alien species is also essential, as their presence can significantly disrupt native community composition and ecosystem functioning (Vujanović et al. 2022; Gallardo et al. 2024). Despite its prevalence, species richness alone is a limited proxy, as it does not necessarily reflect ecosystem functioning. Increasing species numbers does not guarantee better ecological health and, on the contrary, ecosystems may maintain functionality despite species loss—up to a threshold—before collapsing (the "rivet" hypothesis) (Nobles 2025).

Likewise, the use of lists of threatened or protected species remains a widely accepted and valuable tool in conservation planning. While such lists may sometimes be based on incomplete or outdated data and may not fully account for emerging threats, they nonetheless represent a synthesis of current scientific knowledge and societal values regarding biodiversity. These lists help guide conservation priorities and policy actions, reflecting collective judgments about which species warrant particular attention. Still, caution is warranted, as conservation efforts can be unevenly directed toward charismatic or visually appealing taxa, potentially overlooking less conspicuous but ecologically vital groups such as insects, fungi, or microorganisms, which remain underrepresented despite their fundamental roles in ecosystem functioning (Nobles 2025).

Citizen science has become increasingly prominent, with widespread public involvement in nature observations contributing to large-scale biodiversity monitoring initiatives (Callaghan et al. 2021; Macgregor et al. 2021). It represents a valuable tool for gathering species occurrence data, especially in under-sampled or overlooked areas. Therefore, the use of such data for large-scale site sampling at the regional level appears fully relevant. However, although this paper does not aim to assess the quality of the data used, it is important to acknowledge that the associated biases are well-documented in the literature. Citizen science data are not without constraints: sampling effort is often uneven, observer expertise varies, and absence data are typically lacking (Mair & Ruete 2016; Callaghan et al. 2019). These datasets are usually presence-only. Over the past decade, citizen science initiatives have been increasingly supported by on-line platforms, which facilitate data collection and sharing. However, the resulting data are not always aggregated or made openly accessible to the scientific community. As a result, species occurrence data often remain fragmented across multiple databases, hindering large-scale analyses.

In this study, we assess the conservation value of brownfields in post-industrial cities using citizen science data on a wide range of taxa, and considering the limitations linked to this type of dataset. After quantifying the sampling effort across various categories of brownfields, we examine species richness and the presence of both protected and invasive species within these sites. We also compare these conservation value metrics between brownfields and other urban land use types.

Material and methods

Study area and site selection

The study area is situated within 3 post-industrial urban areas of Belgium: Mons, Charleroi, and Liege (Fig. 1). Characterized by a heavy industrial activity in the 19th and 20th centuries, these urban areas inherited numerous brownfields following the coal crisis of 1974, which led to a major shutdown of coal and metallurgy industries (Vandermotten 1998). The three densely urbanized post-industrial areas—Mons, Charleroi, and Liège—cover 25,238 ha, 20,047 ha, and 36,592 ha respectively. These figures correspond to morphological urban areas, defined by the continuity of the built-up area rather than administrative boundaries. These urban areas are situated in Wallonia, a region of southern Belgium characterized by a relatively high population density (218,5 inhabitants per km²) according to data from the Walloon Institute for Evaluation, Foresight and Statistics (IWEPS, version dated 01/06/2025).

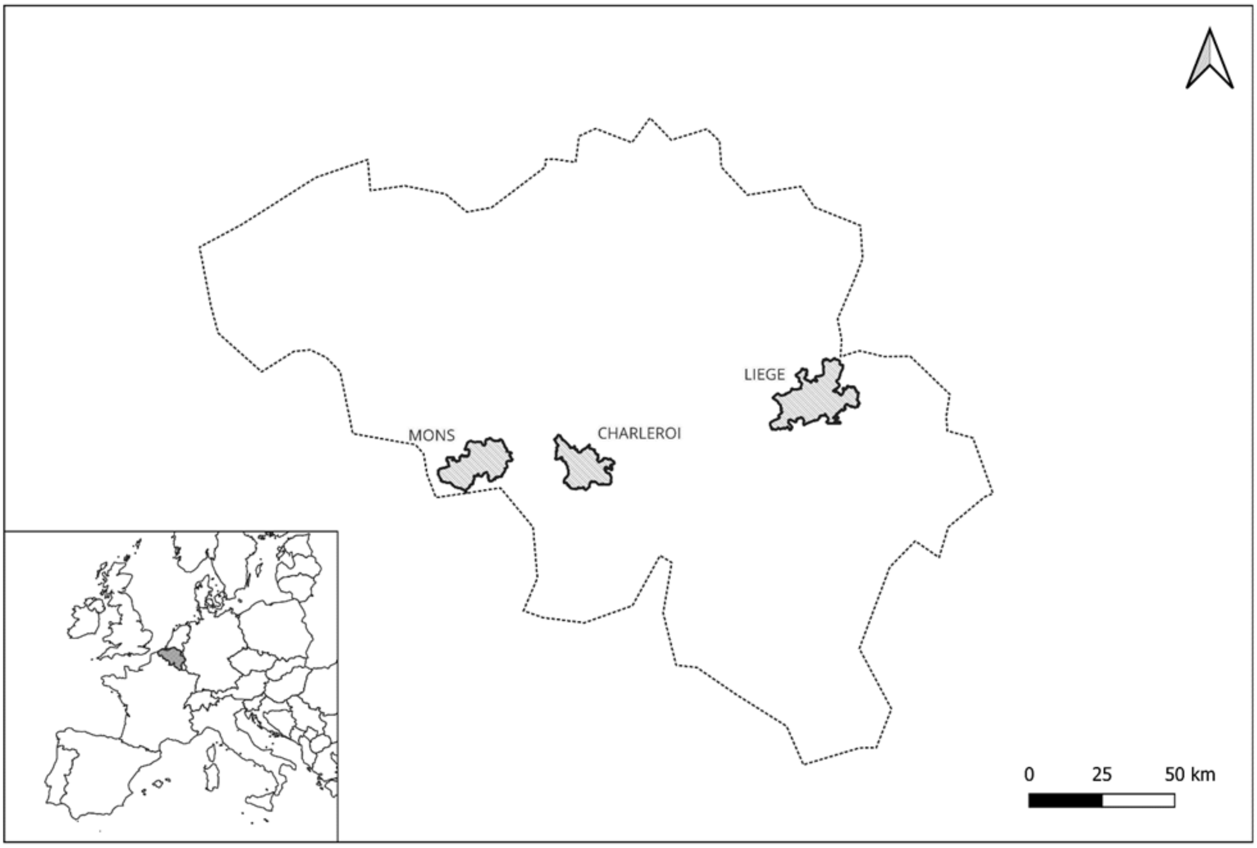

Fig. 1 - Locations of study areas in Belgium

Brownfield were identified using a regional database listing sites characterized by a history of industrial activity and potential for rehabilitation. We selected sites that met the following criteria: a minimum area of 1 hectare, at least five years of inactivity, and an industrial past. In total, 434 sites were selected, including 110 in Mons, 175 in

Charleroi, and 149 in Liège. Most of those sites ceased their activity at the end of the 20th century. These sites represent approximately 0,27% of the urban area in Mons, 1,6% in Liège, and 7,6% in Charleroi.
Selected brownfields encompass 56% forest habitat (including both broadleaved and coniferous trees, ranging in height below or above 3 meters), 26% open habitat (encompassing bare soils, areas with humid and relatively dry soils, permanent monospecific graminoid covers, ploughed herbaceous covers, and disturbed open area), 17% impervious areas and 1% water bodies (Radoux et al., 2023).

The 434 selected sites were attributed to six categories of brownfields, according to the information available in the database: 1) coal mine sites; 2) quarries; 3) heavy industries; 4) disposal sites; 5) transport infrastructures; and 6) other economic activities. Coal mine sites include the places of extraction as well as the slag heap, both related to coal mining. Quarries are sites where mineral extraction took place, as well as their sorting area. Heavy industries include production of material goods other than those related to the mineral extraction. The nature of the activities is various, including metallurgy, textile manufacturing, and chemical industry. Disposal sites are sites known to be used (legally or illegally) as areas of open dumping of garbage, inert materials, and sludge deposit. Transport infrastructures include sites only used for transport, such as freight yards, train stations, repair facility or yard limits. The sites that could not fit within one of the five previous categories were included in other economic activities.

Species occurrence data

This study relies on the two most comprehensive citizen science databases available at the regional level: Observations.be and OFFH (Observatory of Fauna, Flora, and Habitats). Observations.be is a widely used and highly popular platform in the region for recording species occurrences. It is part of a broader European network and is managed by major regional NGOs (Natuurpunt, Natagora) and the International Observation Foundation. OFFH is a web-based portal developed by the Regional Research Department for Nature and Agricultural Areas (DEMNA), part of the Walloon Natural and Agricultural Environmental Studies Department (SPW-ARNE). This latter source also incorporates data from scientific monitoring and research projects.

Species occurrence data from both platforms were merged. We defined a record following Vallet et al. (2012) as a species observed at a specific location and date by a given observer. Duplicate records—those sharing the same species id, observer, date, and location—were excluded, as were records with spatial uncertainty exceeding 100 meters. To ensure temporal relevance, only data collected between January 1, 2000 and December 1, 2022 were considered. Additionally, records from non-target biological kingdoms (Bacteria, Protozoa, Chromista, and Fungi) were excluded, resulting in a final dataset of 60725 records.

To standardize taxonomy, we used the standardize_taxa function from the traitdataform R package (Schneider et al., 2019) under R version 4.3.1. Accepted species names were retrieved and matched to their original entries using the GBIF Backbone Taxonomy. This taxonomy also allowed for the aggregation of species into higher taxonomic groups (Class level or above), based on phylogenetic proximity, data availability, and the categories used in legal designations of protected species. The following groups were used: Amphibia, Arachnida, Aves, Insecta, Mammalia, Plants, Reptilia, and Others. The "Other" category includes Bivalvia, Chilopoda, Clitellata, Diplopoda, Gastropoda, Malacostraca, and Actinopterygii. The two classes Squamata and Testudines were grouped under Reptilia.

Data analysis

Species were assigned a protection or invasive status based on current legal frameworks. Protection status was determined according to the Belgian Nature Conservation Act (NCA) of 1973. We compiled a list of protected animal species present in Wallonia and listed in Annexes 1, 2a, 2b, and 3 of the NCA. Since birds are governed by a distinct legal provision—Article 2—separate from the annexes used for other taxonomic groups, they were treated independently when assigning protection status, to reflect this specific legal framework. For Hymenoptera, five genera—Coelioxys, Epeolus, Eucera, Macropis, and Panurgus—are strictly protected under the NCA. Species from these genera recorded in Belgium via GBIF were added to the list. A list of protected plant species was also established based on Annexes 6a, 6b, and 7 of the NCA. All species within the Bryophyta class are protected under Annex 7. The total number of Bryophyta species considered in this study (498) corresponds to those listed in the Atlas of Walloon Bryophyta (Sotiaux & Vanderpoorten 2015).

Regarding invasive species, 47 animal species were identified as invasive based on their inclusion in the list of "species of Union concern" under EU Regulation 1143/2014. For plants, we referred to the regional decree of September 15, 2022, which lists both species of Union concern and additional regionally recognized invasive species, amounting to a total of 43 invasive plant species.

To assess the sampling effort across various categories of brownfields, we extracted species occurrence data located within the selected brownfield sites. For each brownfield category, we compared: (1) the number and proportion of sites with and without biological records, (2) the total surface area of the sites, (3) the number of records, and (4) the distribution of records per site. Sampling effort across brownfield categories was assessed using an index defined as the ratio of the percentage of total species records to the percentage of total surface area represented by each category (%data / %surface). A ratio close to 1 was interpreted as indicating a well-sampled category.

To address our objective of examining species richness and the presence of both protected and invasive species within these sites, we analysed, for each selected group of fauna and flora, the total number of observed species as well as the proportion of Walloon protected, and invasive species recorded in brownfields. Additionally, we examined the number of protected and invasive species per site, and the proportion of brownfields that hosted at least one protected or invasive species.

Our objective consisting in comparing these conservation value metrics between brownfields and other urban land use types involved a comparative analysis with four additional land use categories: (1) Agriculture, (2) Forest, (3) Residential, and (4) Green Spaces. These categories were defined based on the OpenStreetMap land use map (version dated 19/08/2024). The "Agriculture" category includes areas labelled as meadow, orchard, vineyard, or farmland. "Forest" corresponds to areas designated as forest. "Green Spaces" include parks and recreational grounds, while "Residential" pertains to areas marked as residential.

A 20 km zone was delineated around the selected urban areas, and land use within these urban areas and surrounding buffers was analysed. This buffer distance was chosen to ensure that all land uses considered were embedded within a broadly similar urban context, likely subject to comparable environmental conditions and sharing the same regional species pool. For each land use category, including brownfields, 100 points were randomly distributed. A 125-meter radius buffer—corresponding to the median brownfield size—was generated around each point. Buffers were retained only if they met the following criteria: (1) no spatial overlap with other buffers, and (2) at least 80% of their surface area falling within the designated land use category.

The random distribution of points and buffers generation process were repeated ten times to account for spatial variability and ensure robust results. Species occurrence records were then intersected with the buffers generated across all iterations.

We subsequently analysed the following metrics across land use types: the mean number of buffers containing biological records, the mean number of protected and invasive species per buffer, the relative species count per category (number of species divided by the number of observations), and the mean number of buffers hosting at least one protected or invasive species. Differences across land uses were tested using ANOVA. Post hoc comparisons were performed using Tukey's test. The relationship between mean species records density (number of data points per hectare) and land use type and the comparison of relative species count per land use type were assessed using a Kruskal-Wallis test and a post hoc comparison was performed using Conover's test. To improve normality, the number of protected species was square root-transformed.

Results

Sampling effort in brownfields

The number of sites varied considerably across brownfields categories (Fig.2a). Former coal mine sites accounted for nearly half of the inventoried brownfields. These were followed by industrial sites, which comprised 109 locations, representing 25% of the total.

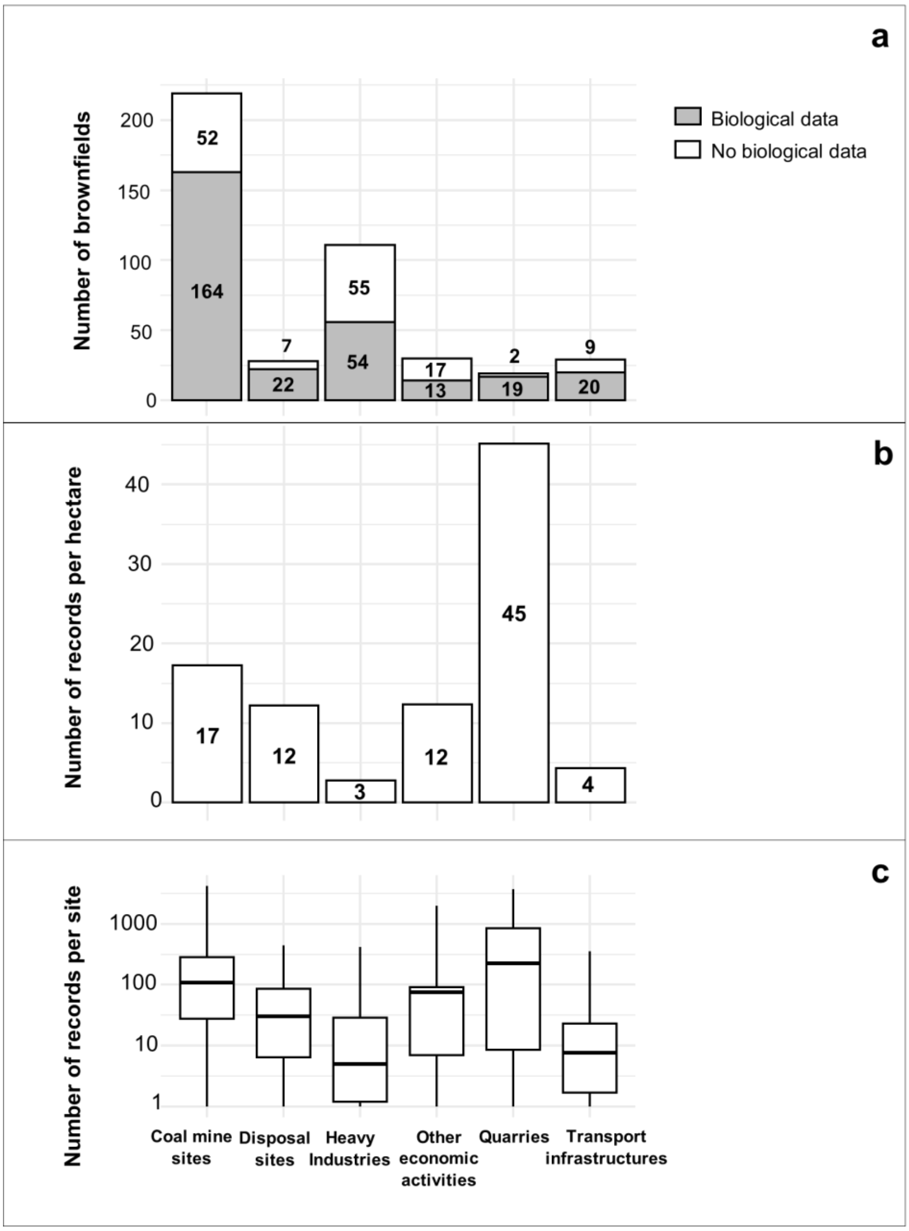

Fig. 2 – Number of sites with and without data per brownfield category (**a**), total number of records per hectare for each brownfield category (**b**) boxplots of the number of records per site for each brownfield category (**c**)

The brownfield categories with the highest number of biological records were quarries, 90,5 % of which contained biological data (Fig. 2a). These were followed by coal mine sites and disposal sites, both of which showed 75,9% of sites with associated biological data.

The amount of data per hectare (Fig. 2b) also varied across brownfield categories. Quarries had the highest ratio (45), followed by coal mines (17), disposal sites (12) and other economic activities (12). Industries (3) and transport infrastructures (4) had lower values.

The number of records per site (Fig. 2c) varied across brownfield categories. Quarries (n = 225) had the highest median value, with a wide interquartile range indicating higher variability, followed by coal mines (n = 109). In contrast, industries (n = 5) showed the narrowest range and lower median values.

The distribution of biological data across brownfield sites was highly uneven (Fig. 2). A small number of sites — mainly former coal mining areas — concentrated the majority of records, with over one-third of all data coming from the ten most surveyed sites. In total, 75% of the data were collected in just 40% of the inventoried brownfields. Conversely, 33% of sites lacked any biological information. Specifically, biological data were available for 291 brownfield sites, representing a total area of 3 527,9 ha, while 143 sites — covering 514,3 ha — had no biological data.

There was also a disparity in the sampling effort across brownfield categories, with former coal mining sites and quarries being overrepresented. In fact, quarries exhibited the highest sampling effort index (3,02), suggesting a substantial oversampling relative to their surface area. Coal mine sites (1,2) and other economic activities (0,81) displayed ratios close to 1, reflecting a balanced sampling relative to their surface areas. Transport infrastructures (0,28) and heavy industries (0,19) had the lowest ratios indicating underrepresentation in the sampling effort.

Analysis of species occurrence data in brownfields.

Plants and insects constituted the largest proportions of records, accounting for 44% and 41% of the data, respectively (Fig.3). The remaining taxonomic groups (Aves, Amphibia, and Mammalia) contributed 7%, 4%, and 2%, respectively. A total of 2361 animal species and 1077 plants species have been recorded (Tab. 1).

Tab. 1 - Number and percentage of species with conservation status or regulatory classification, by taxonomic group, within the Animalia and Plantae kingdoms, observed in brownfields. The proportion of protected species corresponds to the number of protected species observed in brownfields divided by the total number of protected species listed in the CAN. The number of IAS corresponds to the number of invasive species observed in brownfields divided by the total number of invasive species listed at the European or regional level

Taxonomic group	**Total number of species**	**Number of protected species**	**Proportion of protected species**	**Number of invasive species**	**Proportion of invasive species**
Mammalia	41	21	50,00%	3	23,08%
Aves	141	110	73,83%	1	20,00%
Reptile	7	5	71,43%	1	100,00%
Amphibia	13	11	64,71%	1	33,33%
Insecta	1982	43	24,02%	1	20,00%
Arachnida	85	0	0,00%	0	0,00%
Others	92	0	0,00%	1	5,00%
Plants	1077	122	14,68%	33	76,74%

The analysis reveals clear differences between taxonomic groups in both species richness and the proportion of protected and invasive species. Insecta was the most species-rich group, with 1 982 species recorded, followed by Plantae (1 077), Aves (141), and Arachnida (85). Other groups such as Mammalia (41), Amphibia (13), and Reptilia (7) showed lower species counts.

As shown in Table 1, in terms of conservation status, Aves showed the highest proportion of protected species (73,8%), followed by Reptilia (71,4%), Amphibia (64,7%), and Mammalia (50%). This indicates that most species listed as protected are found in brownfields. For Plantae, 122 species were listed as protected, representing 11,3% of the plant species recorded. In contrast, only 2,2% of the insect species were protected, and no protected species were recorded among Arachnida or the group labeled "Others." Overall, a total of 323 protected species were observed and protected species were observed in 185 of the 291 brownfield sites for which biological data were available.

Invasive species were also recorded across several groups. The highest number and proportion were found in Plantae, with 33 invasive species representing 76,7% of the regionally listed invasive plants. Lower proportions of invasive species were observed in Mammalia (23,1%), Amphibia (33,3%), and Insecta (0,05%).

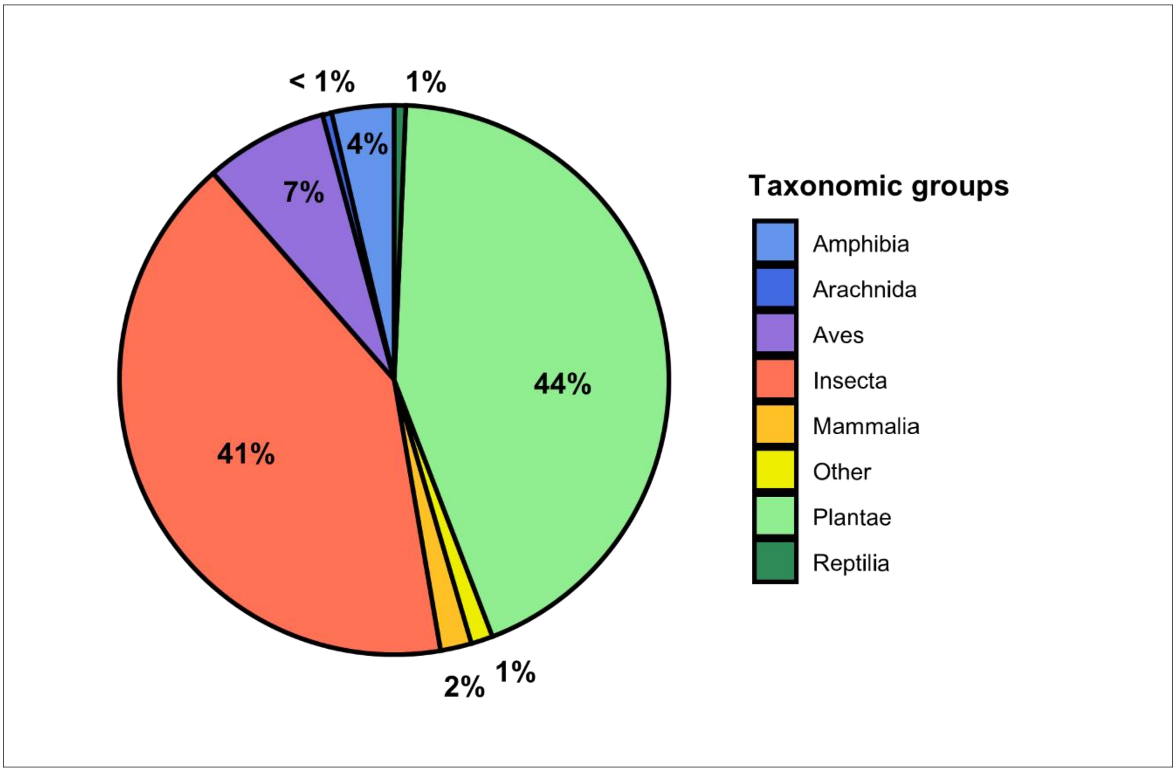

Fig.3 - Proportion of records in the different groups within Animalia and Plantae kingdom. The total number of records is 60725

The 10 most frequently observed protected bird species were : *Phylloscopus collybita* (observed in 27.7% of the study sites containing fauna data), *Parus major* (26,9%), *Sylvia atricapilla* (26,0%), *Turdus merula* (26,0%), *Corvus corone* (25,2%), *Pica pica* (24,4%), *Cyanistes caeruleus* (24,0%), *Erithacus rubecula* (24,0%), *Garrulus glandarius* (23,6%) and *Troglodytes troglodytes* (22,7%).

The 10 most frequently observed protected animal species, excluding birds, were: *Oedipoda caerulescens* (observed in 35,5% of the study sites containing fauna data), *Epidalea calamita* (26%), *Euplagia quadripunctaria* (22,7%), *Rana temporaria* (19,8%), *Ichthyosaura alpestris* (18,2%), *Anguis fragilis* (17,8%), *Bufo bufo* (14,5%), *Alytes obstetricans* (11,2%), *Lissotriton vulgaris* (10,7%) and *Sympecma fusca* (10,3%).

The 10 most frequently observed protected plant species were: *Epipactis helleborine* (observed in 29% of the study sites containing plant data), *Centaurium erythraea* (16,6%), *Taxus baccata* (8,9%), *Ophrys apifera* (7,7%), *Hyacinthoides non-scripta* (4,6%), *Centaurium pulchellum* (4,6%), *Rosa rubiginosa* (4,2%), *Anacamptis pyramidalis* (3,1%), *Neottia ovata* (3,1%), and *Galanthus nivalis* (2,3%).

Comparison of sampling effort and species occurrence data between brownfields and other land uses

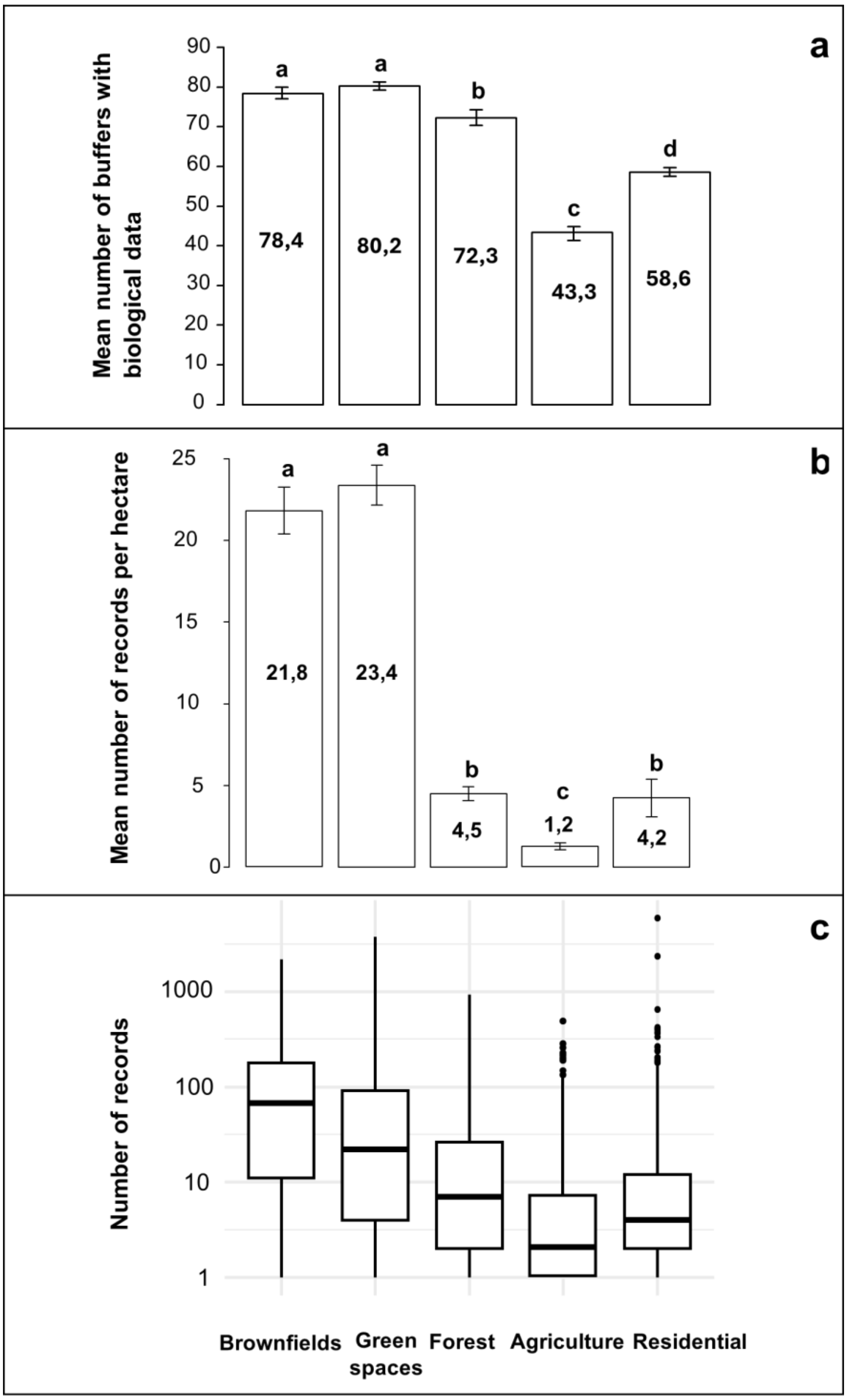

Fig. 4 - Number of buffers with data per land use category. Means were compared using ANOVA, followed by Tukey's post hoc test (**a**), total number of records per hectare for each land use category Means were compared using the Kruskal–Wallis test, followed by Conover's post hoc test. (**b**) and boxplots of the number of records per buffer (**c**) for each land use category. Error bars represent standard errors

The mean number of buffers with biological data (Fig. 4a) differed significantly across land-use types (ANOVA, $F_{4, 45}$ = 105,8, p < 0,001). The highest values were observed in green spaces (80,2 ± 1,0) and brownfields (78,4 ±

1,4). Forest (72,3 ± 2,0), residential areas, (58,6 ± 1,1), and agriculture (43,3 ± 1,7) exhibited significantly different values.

As indicated in the Figure 4b, the mean number of records per hectare also varied significantly across land-use types (Kruskal-Wallis, df=4, $p < 0,05$). Green spaces (23.4 ± 1,2) and brownfields (21,8 ± 1,4) had the highest vales. Forest (4,5 ± 0,4) and residential (4,2 ± 1,1) had lower means but did not significantly differ from each other and agriculture exhibited the lowest mean (1,2 ± 0,2).

The mean number of buffers containing protected species (Fig. 5c) also differed significantly across land uses (ANOVA, $F_{4, 45} = 127,9$, $p < 0,001$). Brownfields (67,9 ± 1,4) and green spaces (64,1 ± 1,7) showed the highest means. Agricultural areas (35,6 ± 1,5) and residential areas (32,5 ± 1,0) did not significantly differ from each other and had lower means.

As underlined in the Figure 5a, statistical analysis highlighted significant differences in the mean of number of observed protected species per buffer across land-use types (ANOVA, $F_{4, 45} = 104,6$, $p < 0,001$). The highest means was recorded in brownfields (160,5 ± 7,9) and green spaces (150,2 ± 3,5). Residential areas (57,4 ± 3,1) and forests (71,2 ± 6,0) had similarly lower means while agriculture exhibited the lowest mean number of protected species (37,4 ± 4,6).

The relative species count significantly differed among land use categories (Kruskal-Wallis, df = 4, $p < 0,001$). On average, agricultural (0,41 ± 0,033) and residential areas, (0,34 ± 0,044) showed similar values and the highest relative numbers of species while the lowest values were recorded in green spaces (0,109 ± 0,0066) and brownfields (0,16 ± 0,0045). Forests exhibited intermediate value (0,29 ± 0,013) but did not differ from residential areas.

Overall, the taxonomic composition of protected species (Fig. 5a) showed consistent patterns across land-use types, with birds and plants dominating in most cases. In green spaces, residential areas, brownfields and forests, these two groups accounted for the majority of protected species richness, with birds representing between 45% and 53% of the total. However, this pattern was more pronounced in agricultural areas, where other taxonomic groups were scarcely represented or even absent, resulting in birds comprising up to 87% of the total protected species.

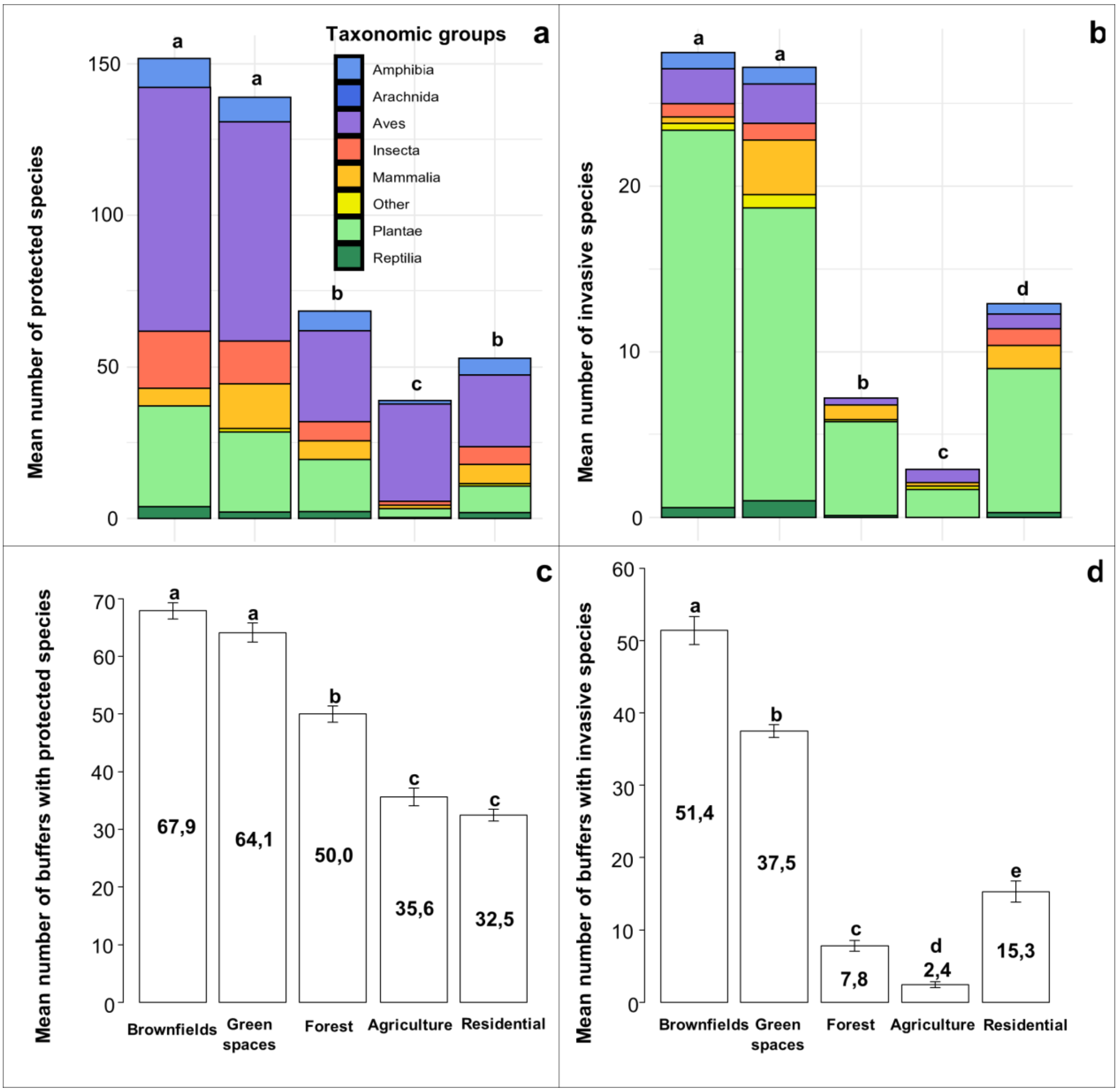

Fig. 5 – Mean number of protected species (**a**), mean number of invasive species (**b**), mean number of buffers with protected species (**c**), and mean number of buffers with invasives species (**d**), for each land use category. Mean values were compared using an ANOVA, and pairwise differences were assessed with Tukey's post hoc test. Error bars represent standard errors

The mean number of buffers with invasive species (Fig. 5d) also differed significantly across land uses (ANOVA, $F_{4, 45} = 277{,}2$, $p < 0{,}001$). Brownfields (51,4 ± 1,9) had the highest values, followed by green spaces (37,5 ± 0,9), residential areas (15,3 ± 1,5), forests (7,8 ± 0,8), and agricultural areas (2,4 ± 0,4).

The mean number of invasive species per buffer varied significantly across land-use types (Fig. 5b; ANOVA, $F_{4, 45} = 153{,}5$, $p < 0{,}001$). The highest means were observed in brownfields (28,1 ± 0,9) and green spaces (27,1 ± 0,9). Residential areas (12,9 ± 1,2), forests (7,2 ± 0,9), and agricultural areas (2.7 ± 0,6) exhibited lower means, significantly different from each other.

Plants dominated all land uses, consistently accounting for the largest share of invasive species richness. (Fig. 5b).

Discussion

Citizen science data revealed a strong potential of brownfield for biodiversity conservation in urban post-industrial environment. Existing data showed a high percentage of sampled brownfields with protected species, remarkably amphibians. Among all land uses, brownfields and green spaces exhibited the highest occurrence of protected species, particularly among plants and birds.

Simultaneously, our study revealed substantial variation in sampling effort and the distribution of biological data across brownfield categories, with former coal-related sites and quarries being the most extensively surveyed. Sampling effort was notably uneven, with certain categories—such as quarries—disproportionately sampled relative to their surface area. Notably, a small number of sites concentrate the majority of biological records, while many remain poorly documented or entirely devoid of data. Sampling bias may stem from citizens' perceptions — positive or negative — of specific brownfield types, as well as from physical access limitations, such as restricted access to private land (Cox et al. 2023; Cooper et al. 2024).

Observers do not inventory sites in a uniform manner and often exhibit specific preferences (Boakes et al. 2016). Site accessibility and the presence of visitor-friendly facilities tend to increase a site's attractiveness, thereby influencing survey frequency. Likewise, not all habitats are equally covered: ornithologists, for instance, tend to favor sites with water bodies, which facilitate bird observation and often support greater avian diversity (Boakes et al. 2016). Species themselves are also subject to unequal recording effort. Additionally, differences in the detectability and identification difficulty of various taxonomic groups must be considered. These factors result in heterogeneous data quantities across taxa. For example, plant species are generally easier to record in large numbers due to their stationary nature and higher visibility, in contrast to more elusive groups such as reptiles. Those that are easier to identify are more likely to be reported. This trend is also evident in our data, where 44% of the encoded records concern plants—the most frequently inventoried taxonomic group—while reptiles account for only 1% of the data. Moreover, species that receive substantial media attention—such as invasive alien species—are often recorded more frequently, partly due to heightened public awareness and prevention campaigns (Boakes et al. 2016).

This uneven sampling effort raises concerns regarding the reliability of ecological assessments and the robustness of conservation-related decision-making based only on existing citizen science data. Specifically, industrial brownfields are the least inventoried in our dataset, which likely contributes to an underestimation of their conservation value (Cox et al. 2023). Despite these disparities, our findings are consistent with previous studies that highlight the high conservation potential of brownfields (Macgregor et al. 2022).

Based on the most frequently observed protected species – such as *Oedipoda caerulescens*, *Epidalea calamita*, *Euplagia quadripunctaria* among fauna and *Centaurium erythraea*, *Ophrys apifera* among flora - our results suggest that pioneer and early succession habitat such as grassland play a crucial role in supporting rare species within urban areas. Brownfield meadows, including grasslands on former landfill sites, have been shown to host species-rich communities comparable to those found in natural or semi-natural habitats (Tarrant et al. 2012; Rahman et al. 2015). Additionally, aquatic habitats found in brownfields, such as temporary ponds, are valuable for biodiversity conservation given the high abundance and diversity of amphibian species they support. These findings reinforce the idea that brownfields can act as substitute habitats for regionally threatened species and underscores their conservation value. Given that brownfields represent a significant proportion of the studied territories—up to 7.6% in some urban areas like Charleroi—and host numerous pioneer species, it would be valuable to highlight their potential role in the development of ecological networks. The high occurrence of these pioneer species across a substantial area of brownfields suggests that these sites could serve as important stepping stones or corridors, facilitating connectivity and supporting the persistence of early-succession species within urban landscapes.

However, among industrial brownfields, some sites exhibit high levels of contamination, notably with heavy metals. It has been demonstrated that such pollution can adversely affect certain species, such as pollinators, whose reproduction is impaired when exposed to these contaminants (Scott et al. 2023). Consequently, this could represent a limiting factor that reduces the potential benefits of brownfields for biodiversity.

Our results also confirm that brownfields harbour a significant number of invasive alien species — a trend previously noted by several authors, who found that invasive plants can dominate brownfield communities and represent a substantial share of overall species richness (Godefroid et al. 2007, Lemoine 2016). The prevalence of invasive species was shown to be influenced by the urbanization gradient, with higher urbanization levels promoting invasions (Penone et al. 2012). As our study sites are in densely urbanized areas, the observed levels of invasiveness likely reflect this context.

In urban green spaces, public perceptions of native or indigenous plant species are shaped by multiple factors, including age, gender, cultural background, and education level (Russo et al. 2025). However, broader public preferences are often driven primarily by aesthetic qualities and by the provision of ecosystem services such as temperature regulation and air pollution mitigation, which frequently outweigh considerations of nativeness. Furthermore, familiarity with a species through frequent exposure, or its emotional appeal («cuddliness»), can reduce concerns about its non-native status and lead to stronger opposition to its removal or eradication (Russo et al. 2025).

Non-native plant species, despite their ecological risks, can offer significant benefits in urban ecosystems. They contribute to practices such as urban foraging, which fosters social cohesion and strengthens human-nature connections. Consequently, recent studies recommend moving beyond a strict native-versus-non-native framework to emphasize species diversity and functional traits, such as resilience and the provision of ecosystem services (Russo et al. 2025).

Nevertheless, caution is warranted as certain non-native species are invasive, causing ecosystem disruptions through altered biotic interactions, economic costs, and adverse human health effects (Russo et al. 2025). Research on urban pollinators also highlights a marked preference for native perennial plants, underscoring the importance of native flora for sustaining urban insect biodiversity (Russo et al. 2025). In our study, we focused on the subset of invasive species listed in European or regional regulations, which include species with well-documented and significant negative impacts.

The comparison between brownfields and other land-use types revealed a strong similarity with green spaces in terms of the number of protected and invasive species. This aligns with Lemoine's (2016) conclusions on ecological roles of urban green areas, which support significant biodiversity levels due to their specific species assemblages resulting from varying levels of management. While green spaces and brownfields are complementary, the latter often display greater variability in vegetation stages, even in the absence of management, due to a heterogeneity of abiotic conditions. As Barker (2015) noted, even relatively young brownfields can develop conservation values approaching those of long-established forests. In contrast, residential and agricultural areas exhibited lower conservation value, likely due to the intensity of human activities and associated ecological disturbances (McKinney & Michael 2008; Dallimer et al. 2012; Reis et al. 2012; Estrada-Carmona et al. 2022).

The NCA listings provide a standardized and clear inventory of protected species; however, it may overlook rare or underrepresented taxa, potentially leading to an underestimation of site conservation value. This pattern is particularly evident in the Walloon context, where some taxonomic groups—such as Arachnida—are poorly represented in legal protection frameworks, often due to limited data availability or insufficient knowledge about their distribution and ecology. As a result, these groups are frequently excluded from conservation assessments despite their ecological importance. Conversely, other groups like birds are overrepresented, with the majority of species receiving protection, including many that are common and widespread. This disproportionate representation can create a misleading impression of high conservation value, when in fact it may reflect ordinary species communities rather than rare or threatened ones.

In addition, the use of relative species richness accounts for the correlation between the number of species recorded and the total number of observations, thereby offering a normalized measure of biodiversity. However, the relevance of this correction diminishes when the same species are repeatedly recorded, which may be the case for certain taxonomic groups. This limitation is less critical for taxa such as plants or insects, which encompass a large number of species, but may be a problem for groups with inherently low species richness, such as reptiles or amphibians. Consequently, sites with a high number of observations for these groups may appear less biodiverse under this metric, despite potentially serving as important refuges for them.

Contrarily to vacant lots and other ephemeral areas in urban areas, post-industrial brownfields, especially those affected by contamination, tend to be relatively stable over time. Since industrial activities have ceased and are unlikely to resume in the coming decades, these sites are less subject to urban redevelopment dynamics. As such, they may be seen not as temporary land-use gaps but as potential long-term urban natural areas. Depending on their shape, size, and spatial configuration, brownfields can play various roles within the urban ecological network, functioning as core habitats, stepping stones, or ecological corridors that facilitate species movement across fragmented urban landscapes. Among these, transportation-related brownfields are particularly important, as their typically linear and continuous structure makes them especially well-suited to serve as ecological corridors. By fulfilling this role, they can significantly enhance ecological connectivity, support urban wildlife movement, and contribute to broader green infrastructure networks (Kim et al. 2018). A diverse mosaic of brownfields varying in size and successional stage has been shown to maximize biodiversity (Kattwinkel et al. 2009; Kattwinkel et al. 2011; Bonthoux et al. 2014). Beyond their role as ecological connectors, industrial brownfields could also provide key ecosystem services. Their forest structure could support stormwater management by reducing runoff and aiding groundwater recharge or even contribute to carbon sequestration and climate regulation, offering lasting environmental benefits in urban settings (Kim et al. 2018). Despite each site has its own conservation value urban biodiversity conservation must be approached at the territorial scale – not just through isolated site-by-site interventions (Kattwinkel et al. 2011).

Despite their ecological potential, brownfields are frequently associated with negative perceptions, which can contribute to their marginalization in urban conservation policies. This societal view tends to prioritize economic redevelopment over ecological considerations (Cooper et al. 2024). Urban planning decisions — often shaping public sentiment — may favor redevelopment of former industrial sites rather than infringing on existing green spaces. However, our results indicate that such redevelopment may threaten a diversity of protected species that find suitable habitat in brownfields. While our study focused on three post-industrial cities, and thus a primarily urbanized landscape, natural areas within this region are likely influenced by the surrounding urban context. Broadening the spatial scale to include less urbanized landscapes could yield complementary conclusions and uncover greater biodiversity in other land-use categories. Future research should therefore adopt a larger regional or national scope, incorporate transitional habitat types, and address the challenge of sampling less accessible sites. These steps will help build a more complete and reliable understanding of the conservation value of urban and peri-urban ecosystems.

Acknowledgements

We gratefully acknowledge J. Teller and J. Jacquemin from the ArGEnCo department at the University of Liège for their valuable support in mapping the brownfield sites within our study area. We also extend our sincere thanks to the Observations.be platform (managed by the NGO Natagora) and the Walloon Department for the Study of the Natural and Agricultural Environment (DEMNA) for providing open-access biodiversity data.

References

Aronson, M. F., 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

Bernard, J. (2012). *Friches urbaines et biodiversité—Rencontres de Natureparif*.

Boakes, E. H., Gliozzo, G., Seymour, V., Harvey, M., Smith, C., Roy, D. B., & Haklay, M. (2016). Patterns of contribution to citizen science biodiversity projects increase understanding of volunteers' recording behaviour. *Scientific Reports*, *6*(1), 33051. https://doi.org/10.1038/srep33051

Bonthoux, S., Brun, M., Di Pietro, F., Greulich, S., & Bouché-Pillon, S. (2014). How can wastelands promote biodiversity in cities? A review. *Landscape and Urban Planning*, *132*, 79-88. https://doi.org/10.1016/j.landurbplan.2014.08.010

Broughton, R. K., Parry, W., & Maziarz, M. (2020). Wilding of a post-industrial site provides a habitat refuge for an endangered woodland songbird, the British Willow Tit *Poecile montanus kleinschmidti*. *Bird Study*, *67*(3), 269-278. https://doi.org/10.1080/00063657.2020.1863333

Brouwer, F. M., & Chadwick, M. J. (1991). Future Land Use Patterns in Europe. In F. M. Brouwer, A. J. Thomas, & M. J. Chadwick (Éds.), *Land Use Changes in Europe : Processes of Change, Environmental Transformations and Future Patterns* (p. 49-78). Springer Netherlands. https://doi.org/10.1007/978-94-011-3290-9_3

Callaghan, C. T., Poore, A. G. B., Mesaglio, T., Moles, A. T., Nakagawa, S., Roberts, C., Rowley, J. J. L., VergÉs, A., Wilshire, J. H., & Cornwell, W. K. (2021). Three Frontiers for the Future of Biodiversity Research Using Citizen Science Data. *BioScience*, *71*(1), 55-63. https://doi.org/10.1093/biosci/biaa131

Callaghan, C. T., Rowley, J. J. L., Cornwell, W. K., Poore, A. G. B., & Major, R. E. (2019). Improving big citizen science data : Moving beyond haphazard sampling. *PLOS Biology*, *17*(6), e3000357. https://doi.org/10.1371/journal.pbio.3000357

Changes in bird species richness through different levels of urbanization : Implications for biodiversity conservation and garden design in Central Brazil | Request PDF. (s. d.). *ResearchGate*. https://doi.org/10.1016/j.landurbplan.2012.04.009

Cooper, C., Collier, M. J., Pineda-Pinto, M., Castañeda, N. R., O'Donnell, M., & Nulty, F. (2024). Urban novel ecosystems as affective landscapes. *Sustainability Science*, *19*(6), 1921-1933. https://doi.org/10.1007/s11625-024-01539-w

Cox, L., & Rodway-Dyer, S. (2023). The underappreciated value of brownfield sites : Motivations and challenges associated with maintaining biodiversity. *Journal of Environmental Planning and Management*. https://www.tandfonline.com/doi/abs/10.1080/09640568.2022.2050683

Dallimer, M., Rouquette, J. R., Skinner, A. M. J., Armsworth, P. R., Maltby, L. M., Warren, P. H., & Gaston, K. J. (2012). Contrasting patterns in species richness of birds, butterflies and plants along riparian corridors in an urban landscape. *Diversity and Distributions*, *18*(8), 742-753. https://doi.org/10.1111/j.1472-4642.2012.00891.x

Dyer, R. J., Gillings, S., Pywell, R. F., Fox, R., Roy, D. B., & Oliver, T. H. (2017). Developing a biodiversity-based indicator for large-scale environmental assessment : A case study of proposed shale gas extraction sites in Britain. *Journal of Applied Ecology*, *54*(3), 872-882. https://doi.org/10.1111/1365-2664.12784

Estrada-Carmona, N., Sánchez, A. C., Remans, R., & Jones, S. K. (2022). Complex agricultural landscapes host more biodiversity than simple ones : A global meta-analysis. *Proceedings of the National Academy of Sciences*, *119*(38), e2203385119. https://doi.org/10.1073/pnas.2203385119

Fischer, L. K., Von Der Lippe, M., & Kowarik, I. (2013). Urban grassland restoration : Which plant traits make desired species successful colonizers? *Applied Vegetation Science*, *16*(2), 272-285. https://doi.org/10.1111/j.1654-109X.2012.01216.x

Food and Agriculture Organization of the United Nations (FAO), L. and W. (1995). *Planning for sustainable use of land resources : Towards a new approach*. https://openknowledge.fao.org/handle/20.500.14283/v8047e

Gallardo, B., Bacher, S., Barbosa, A. M., Gallien, L., González-Moreno, P., Martínez-Bolea, V., Sorte, C., Vimercati, G., & Vilà, M. (2024). Risks posed by invasive species to the provision of ecosystem services in Europe. *Nature Communications*, *15*(1), 2631. https://doi.org/10.1038/s41467-024-46818-3

Godefroid, S., Monbaliu, D., & Koedam, N. (2007). The role of soil and microclimatic variables in the distribution patterns of urban wasteland flora in Brussels, Belgium. *Landscape and Urban Planning*, *80*(1), 45-55. https://doi.org/10.1016/j.landurbplan.2006.06.001

Higgs, E. S., Harris, J. A., Heger, T., Hobbs, R. J., Murphy, S. D., & Suding, K. N. (2018). Keep ecological restoration open and flexible. *Nature Ecology & Evolution*, *2*(4), 580-580. https://doi.org/10.1038/s41559-018-0483-9

Kamp, J., Pelster, A., Gaedicke, L., Karthäuser, J., Dieker, P., & Mantel, K. (2015). High nest survival and productivity of Northern Lapwings Vanellus vanellus breeding on urban brownfield sites. *Journal of Ornithology*, *156*(1), 179-190. https://doi.org/10.1007/s10336-014-1114-0

Kattwinkel, M., Biedermann, R., & Kleyer, M. (2011). Temporary conservation for urban biodiversity. *Biological Conservation*, *144*(9), 2335-2343. https://doi.org/10.1016/j.biocon.2011.06.012

Kattwinkel, M., Strauss, B., Biedermann, R., & Kleyer, M. (2009). Modelling multi-species response to landscape dynamics : Mosaic cycles support urban biodiversity. *Landscape Ecology*, *24*(7), 929-941. https://doi.org/10.1007/s10980-009-9371-7

Kim, G., Miller, P. A., & Nowak, D. J. (2018). Urban vacant land typology : A tool for managing urban vacant land. *Sustainable Cities and Society*, *36*, 144-156. https://doi.org/10.1016/j.scs.2017.09.014

Kopf, R. K., Finlayson, C. M., Humphries, P., Sims, N. C., & Hladyz, S. (2015). Anthropocene Baselines : Assessing Change and Managing Biodiversity in Human-Dominated Aquatic Ecosystems. *BioScience*, *65*(8), 798-811. https://doi.org/10.1093/biosci/biv092

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

Ksiazek-Mikenas, K., Chaudhary, V. B., Larkin, D. J., & Skogen, K. A. (2021). A habitat analog approach establishes native plant communities on green roofs. Ecosphere, 12(9), e03754. https://doi.org/10.1002/ecs2.3754

Lemoine, G. (2016). Flores et pollinisateurs des villes et des friches urbaines… Entre nature temporaire et biodiversité en mouvement. *Bulletin de la Société Botanique de France*, *69*(1-4), 103-116.

Lizée, M.-H., Mauffrey, J.-F., Tatoni, T., & Deschamps-Cottin, M. (2011). Monitoring urban environments on the basis of biological traits. *Ecological Indicators*, *11*(2), 353-361. https://doi.org/10.1016/j.ecolind.2010.06.003

Loures, L., & Vaz, E. (2018). Exploring expert perception towards brownfield redevelopment benefits according to their typology. Habitat International, 72, 66 76. https://doi.org/10.1016/j.habitatint.2016.11.003

Lundholm, J. T., & Richardson, P. J. (2010). Habitat analogues for reconciliation ecology in urban and industrial environments. *Journal of Applied Ecology*, *47*(5), 966-975.

Macgregor, C. J., Bunting, M. J., Deutz, P., Bourn, N. A. D., Roy, D. B., & Mayes, W. M. (2022). Brownfield sites promote biodiversity at a landscape scale. Science of The Total Environment, 804, 150162. https://doi.org/10.1016/j.scitotenv.2021.150162

Maes, J., Barbosa, A., Baranzelli, C., Zulian, G., Batista e Silva, F., Vandecasteele, I., Hiederer, R., Liquete, C., Paracchini, M. L., Mubareka, S., Jacobs-Crisioni, C., Castillo, C. P., & Lavalle, C. (2015). More green infrastructure is required to maintain ecosystem services under current trends in land-use change in Europe. Landscape Ecology, 30(3), 517 534. https://doi.org/10.1007/s10980-014-0083-2

Mair, L., & Ruete, A. (2016). Explaining Spatial Variation in the Recording Effort of Citizen Science Data across Multiple Taxa. *PLOS ONE*, *11*(1), e0147796. https://doi.org/10.1371/journal.pone.0147796

Martínez-Abraín, A., & Jiménez, J. (2016). Anthropogenic areas as incidental substitutes for original habitat. *Conservation Biology*, *30*(3), 593-598. https://doi.org/10.1111/cobi.12644

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*(3), 247-260. https://doi.org/10.1016/j.biocon.2005.09.005

McKinney, M. L. (2008). Effects of urbanization on species richness : A review of plants and animals. *Urban Ecosystems*, *11*(2), 161-176. https://doi.org/10.1007/s11252-007-0045-4

Merwin, L., Umek, L., & Anastasio, A. E. (2022). Urban post-industrial landscapes have unrealized ecological potential. Restoration Ecology, 30(8), e13643. https://doi.org/10.1111/rec.13643

Muratet, A. (2006). Diversité végétale en milieu urbain : L'exemple des Hauts-de-Seine. Dissertation, University of Paris 6. https://theses.fr/2006PA066206

Muratet, A., Muratet, M., Pellaton, M., Brun, M., Baude, M., Shwartz, A., & Fontaine, C. (2021). Wasteland, a Refuge for Biodiversity, for Humanity. In F. Di Pietro & A. Robert (Éds.), *Urban Wastelands : A Form of Urban Nature?* (p. 95-120). Springer International Publishing. https://doi.org/10.1007/978-3-030-74882-1_5

Nobles, E. C. (2025). The ethical foundations of biodiversity metrics. *Current Opinion in Environmental Sustainability*, *72*, 101503. https://doi.org/10.1016/j.cosust.2024.101503

Nowak, D. J., & Greenfield, E. J. (2020). The increase of impervious cover and decrease of tree cover within urban areas globally (2012–2017). *Urban Forestry & Urban Greening*, *49*, 126638. https://doi.org/10.1016/j.ufug.2020.126638

Observations.be. (s. d.). Observations.be. https://observations.be/stats/ Accessed 29 january 2025

Oke, C., Bekessy, S. A., Frantzeskaki, N., Bush, J., Fitzsimons, J. A., Garrard, G. E., Grenfell, M., Harrison, L., Hartigan, M., Callow, D., Cotter, B., & Gawler, S. (2021). Cities should respond to the biodiversity extinction crisis. Npj Urban Sustainability, 1(1), 1 4. https://doi.org/10.1038/s42949-020-00010-w

Penone, C., Machon, N., Julliard, R., & Le Viol, I. (2012). Do railway edges provide functional connectivity for plant communities in an urban context? Biological Conservation, 148(1), 126 133. https://doi.org/10.1016/j.biocon.2012.01.041

Phillips, D., & Lindquist, M. (2021). Just weeds? Comparing assessed and perceived biodiversity of urban spontaneous vegetation in informal greenspaces in the context of two American legacy cities. *Urban Forestry & Urban Greening*, *62*, 127151. https://doi.org/10.1016/j.ufug.2021.127151

Pocock, M. J. O., Tweddle, J. C., Savage, J., Robinson, L. D., & Roy, H. E. (2017). The diversity and evolution of ecological and environmental citizen science. *PLOS ONE*, *12*(4), e0172579. https://doi.org/10.1371/journal.pone.0172579

Preston, P. D., Dunk, R. M., Smith, G. R., & Cavan, G. (2023). Not all brownfields are equal : A typological assessment reveals hidden green space in the city. *Landscape and Urban Planning*, *229*, 104590. https://doi.org/10.1016/j.landurbplan.2022.104590

Radoux, J., Vroey, M. D., Goffart, B., Coppée, T., Bourdouxhe, A., Dufrêne, M., & Defourny, P. (2023). *Lifewatch-WB geodatabase (v4.17) : Attribute description.*

Rahman, M. L., Tarrant, S., McCollin, D., & Ollerton, J. (2015). Vegetation cover and grasslands in the vicinity accelerate development of carabid beetle assemblages on restored landfill sites. *Zoology and Ecology*, *25*(4), 347-354. https://doi.org/10.1080/21658005.2015.1068521

Reis, E., López-Iborra, G. M., & Pinheiro, R. T. (2012). Changes in bird species richness through different levels of urbanization : Implications for biodiversity conservation and garden design in Central Brazil. *Landscape and Urban Planning*, *107*(1), 31-42. https://doi.org/10.1016/j.landurbplan.2012.04.009

Rupprecht, C. D. D., Byrne, J. A., Garden, J. G., & Hero, J.-M. (2015). Informal urban green space : A trilingual systematic review of its role for biodiversity and trends in the literature. *Urban Forestry & Urban Greening*, *14*(4), 883-908. https://doi.org/10.1016/j.ufug.2015.08.009

Russo, A., Esperon-Rodriguez, M., St-Denis, A., & Tjoelker, M. G. (2025). Native vs. Non-Native Plants : Public Preferences, Ecosystem Services, and Conservation Strategies for Climate-Resilient Urban Green Spaces. *Land*, *14*(5), Article 5. https://doi.org/10.3390/land14050954

Sandlos, J., Wiersma, Y., & Alagona, P. S. (2012). Past Imperfect : Using Historical Ecology and Baseline Data for Contemporary Conservation and Restoration Projects. *Environmental Philosophy*, *9*(1), Article 1.

Schneider, F. D., Fichtmueller, D., Gossner, M. M., Güntsch, A., Jochum, M., König-Ries, B., Le Provost, G., Manning, P., Ostrowski, A., Penone, C., & Simons, N. K. (2019). Towards an ecological trait-data standard. *Methods in Ecology and Evolution*, *10*(12), 2006-2019. https://doi.org/10.1111/2041-210X.13288

Scott, S. B., Sivakoff, F. S., Meuti, M. E., & Gardiner, M. M. (2023). Metals could challenge pollinator conservation in legacy cities. *Journal of Insect Conservation*, *27*(3), 361-375. https://doi.org/10.1007/s10841-023-00474-y

Sotiaux, A., & Vanderpoorten, A. (2015). *Bryophytes de Wallonie. Tome I : Hepatiques. Tome II : Mousses.* https://orbi.uliege.be/handle/2268/206923

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

Tarrant, S., Ollerton, J., Rahman, M. L., Tarrant, J., & McCollin, D. (2013). Grassland Restoration on Landfill Sites in the East Midlands, United Kingdom : An Evaluation of Floral Resources and Pollinating Insects. *Restoration Ecology*, *21*(5), 560-568. https://doi.org/10.1111/j.1526-100X.2012.00942.x

Tomlinson, S., Matthes, U., Richardson, P. J., & Larson, D. W. (2008). The ecological equivalence of quarry floors to alvars. Applied Vegetation Science, 11(1), 73 82. https://doi.org/10.1111/j.1654-109X.2008.tb00206.x

Tureckova, K. (2021). Specific types and categorizations of brownfields: synthesis of individual approaches. Geographia Technica, 16(2), 29 39. https://doi.org/10.21163/GT_2021.162.03

Vallet, J., Rambaud, M., Coquel, L., Poncet, L., & Hendoux, F. (2012). Effort d'échantillonnage et atlas floristiques – exhaustivité des mailles et caractérisation des lacunes dans la connaissance. Comptes Rendus Biologies, 335(12), 753 763. https://doi.org/10.1016/j.crvi.2012.11.005

Vandermotten, C. (1998). Dynamiques spatiales de l'industrialisation et devenir de la Belgique. Le Mouvement social, 185, 75 100. https://doi.org/10.2307/3779048

Vujanović, D., Losapio, G., Milić, S., & Milić, D. (2022). The Impact of Multiple Species Invasion on Soil and Plant Communities Increases With Invasive Species Co-occurrence. Frontiers in Plant Science, 13. https://doi.org/10.3389/fpls.2022.875824

Willis, K. J., Jeffers, E. S., Tovar, C., Long, P. R., Caithness, N., Smit, M. G. D., Hagemann, R., Collin-Hansen, C., & Weissenberger, J. (2012). Determining the ecological value of landscapes beyond protected areas. Biological Conservation, 147(1), 3 12. https://doi.org/10.1016/j.biocon.2011.11.001

Statements & Declarations

Funding

The study was funded by the program ‘Fiche 123 – Plan de Relance de la Wallonie’.

Competing Interests

The authors have no relevant financial and non-financial interests to disclose.

Author Contributions

All authors contributed to the study conception and design. Data analyses were performed by J. Martoglio and B. Cornier. The first draft of the manuscript was written by J. Martoglio and A. Monty and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Data availability

The datasets supporting this study will be made available shortly upon publication or upon reasonable request to the corresponding author.