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RESEARCH ARTICLE

Relative impact of native and non-native plants on endangered species in a major European City

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Abstract

1. International policies increasingly call on decision-makers to enhance urban biodiversity. While cities can support considerable biological richness—including rare and endangered species—the persistence of plant populations in urban habitats is often at risk. Identifying and prioritizing threats to endangered plants is essential for guiding effective conservation strategies.
2. We analysed a unique dataset derived from expert assessments of threats to 1231 populations of 201 endangered species within Berlin's Flora Protection Programme. Threats were categorized and their relative importance quantified at both population and species levels, and across habitat types.
3. Biotic threats were most prevalent—affecting 63% of populations and 78% of species, followed by threats related to agriculture, nutrient inputs, urban development and recreation. Among biotic threats, succession and expansive plant species were similarly important, each affecting 23%–25% of populations, followed by tree and shrub plantings (11.5%). Native species (primarily *Calamagrostis epigejos* and *Phragmites australis*) threatened 16% of populations while non-native species (mainly *Solidago* taxa and *Prunus serotina*) affected 15.2%. The relevance of non-native threats varied across habitat types, with woodlands hosting the largest number of affected populations.
4. Despite Berlin's status as an invasion hotspot, non-native species represented a comparatively minor threat relative to other biotic drivers, agriculture and nutrient inputs—consistently across major habitat types. Notably, no EU-listed Invasive Alien Species were identified as threats to the target taxa—including the widespread *Ailanthus altissima*.
5. *Synthesis and applications*. This first city-wide assessment of threats to multiple populations of endangered plant species provides a foundation for targeted threat management in urban contexts. Maintaining habitat quality and optimizing land use will be most effective in protecting endangered species from competitive pressures. While existing non-native threats should be addressed, their limited role suggests that urban conservation efforts should prioritize more pressing drivers of decline. In Berlin, many endangered native species likely co-exist with

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non-native species at broader spatial scales. Future studies should move beyond isolated assessment of invasion risks, compare the relative importance of different threats and apply this framework across cities in diverse biogeographical and socio-economic settings.

KEYWORDS
biological invasions, causes of decline, conservation management, endangered plants, expansive native species, invasive alien plant species, urban biodiversity conservation

1 | INTRODUCTION

As urbanization continues to accelerate (Simkin et al., 2022), international frameworks such as the Kunming-Montreal Target No. 12 increasingly urge decision-makers—from city to national levels—to reduce pressures on urban green and blue spaces and to enhance urban biodiversity (CBD, 2022; Kowarik et al., 2025). Urban areas can harbour more plant species than less developed areas (Kühn et al., 2004) and may serve as hotspots for endangered species (Ives et al., 2016; Lepczyk et al., 2023), with some cities even hosting the last remaining populations (Soanes & Lentini, 2019). Within cities, a variety of natural and semi-natural habitats can support distinct assemblages of endangered plant species (Knapp et al., 2022; Planchuelo et al., 2019).

However, the contribution of cities to species conservation remains contested (Shwartz et al., 2014). Many species have gone locally extinct in urban areas (Hahs et al., 2009; Duncan et al., 2011), and those that persist often occur at lower abundances compared to historical or non-urban baselines (Aronson et al., 2014; Chocholoušková & Pyšek, 2003). Urban environments pose persistent barriers to recruitment and establishment (Kowarik & von der Lippe, 2018; Piana et al., 2019), as evidenced by a one third loss of endangered species populations over just 7.6 years in Berlin (Planchuelo et al., 2020a). Endangered species in cities are thus burdened by significant extinction debts (Hahs et al., 2009).

Identifying threat factors is therefore key to efficient urban conservation (Burgman et al., 2007; Chamberlain et al., 2016). Previous studies have largely inferred threats indirectly—for example, by correlating species occurrence with environmental variables (e.g. Beninde et al., 2015)—but rarely quantify causal impacts on endangered populations. Trait-based analyses may provide additional insights; for instance, the prevalence of wetland species among local extinctions suggests vulnerability to hydrological changes (Williams et al., 2015). Yet a comprehensive understanding of the diverse threats to urban biodiversity remains limited (Kowarik et al., 2025).

A global review identified land-use change and plant invasions as the most frequently reported threats to urban biodiversity (de Barros Ruas et al., 2022). However, it did not quantify causal relationships or assess the robustness of underlying evidence. While non-native plants can negatively impact urban biodiversity (Gaertner et al., 2017; Potgieter et al., 2017), such effects vary across biogeographic contexts, and empirical evidence for strong invasion impacts in temperate cities is scarce (Gaertner & Kowarik, 2023). Critically, the relative importance of invasion-related threats compared to other threat factors in urban settings remains largely unknown.

To address this gap, we analysed expert-based assessments of threats to 1231 georeferenced populations of 201 endangered plant species in Berlin. Expert elicitation is well established in conservation research (Drescher et al., 2013; Martin et al., 2012), and forms the basis for climate risk assessments (Rastandeh et al., 2018), invasion analyses (Dehnen-Schmutz et al., 2022) and Red List evaluations. However, city-scale threat aggregation may mask site-specific variation and does not permit ranking of threat relevance. Our study aims to determine the relative importance of threats to endangered plant species in Berlin at the local scale, based on expert assessments from the city's Flora Protection Programme (Meißner & Seitz, 2010). Experts recorded all threats for each population using a standardized threat catalogue during field visits.

To inform the ongoing debate on invasion impacts in urban areas (de Barros Ruas et al., 2022; Gaertner et al., 2017; Sax et al., 2022), we specifically analysed the relative importance of expansive native versus non-native plant species as biotic threats. Given the context-dependency of invasion impacts (Catford et al., 2022), we also examined variation in these threats across major habitat types. The following research questions guided our analyses:

1. What is the relative importance of major threat factors to populations of conservation target plant species in Berlin?
2. How do non-native plant species compare with native species and other biotic threats in terms of their relative importance?
3. Does the relevance of non-native species as threat factors differ among major habitat types?

2 | METHODS

2.1 | Study area and study system

Berlin, Germany's capital and largest city (3.9 million inhabitants in 2023), has a temperate climate and is notably green: in 2020, green and blue spaces covered 46.3% of its 892 km^2 area (https://www.berlin.de/umweltatlas/nutzung/flaechennutzung/2020/kartenbeschreibung/). Berlin's flora has been studied for over 200 years (Krausch & Sukopp, 2010) with habitat-scale research dating back

to the 1970s (Kowarik, 2023), underpinning regional Red Lists first issued in 1982 and updated in 1991, 2001 and 2018. The most recent list reports that 17% of Berlin's flora has gone extinct since the mid-19th century, and 29% is currently endangered (Seitz et al., 2018).

Under the Berlin Flora Protection Programme, over 200 priority species have been repeatedly monitored by 30 local experts coordinated by the Berlin Nature Conservation Foundation, the Senate Department for Nature Conservation, and the Botanical Association of Berlin and Brandenburg (Meißner & Seitz, 2010). These target species are critically endangered at the regional, national or international level, or Berlin has a particular responsibility for their conservation (e.g. endemic taxa or species with a global distribution concentrated in Berlin; Meißner & Seitz, 2010; see list in Planchuelo et al., 2020a). Experts georeferenced all known populations of each target species and recorded detailed information on the threat factors affecting them, along with their respective habitat types. In some cases, targeted management interventions were also implemented (Seitz et al., 2018). This has yielded a unique dataset of local populations. Previous analyses have explored their distribution across (semi-) natural and anthropogenic habitats (Planchuelo et al., 2019), their spatial relationship to natural remnants (Planchuelo et al., 2020b) and the probability of local extinctions during the monitoring period (Planchuelo et al., 2020a).

2.2 | Expert assessment of threats

Monitoring experts assessed threats to each population through comprehensive on-site evaluations, drawing on their in-depth local knowledge and expertise in the biology of the endangered plant species. Populations were defined as groups at least 30 m apart or separated by roads or paths. Experts used a pre-structured catalogue from Brandenburg's biotope mapping (LUB, 2019), which lists potential 96 threats—54 of which were recorded in Berlin. We grouped these into eight major categories (Table 1; Table S1 for target species across habitat types), distinguishing those driven by expansive species into native and post-1492 non-native (neophyte) subcategories.

Initially, threats were recorded at the lowest hierarchical level (Table 1), then aggregated so that each higher level category was only counted once per population—avoiding overweighting of related threats (e.g. 'succession' and 'plant competition' together count as a single 'biotic threat'). The 'expansive species' category, encompassing competitive native and non-native species, was quantified separately. Moreover, we determined the numbers of non-native and native species that were listed as threats in the category of 'Tree and shrub plantings'.

2.3 | Statistical analyses

We compared the relative importance of principal threat categories using χ^2 tests, both for the number of species and populations affected. To assess whether populations in different habitat types are disproportionately affected by expansive non-native species, we further subdivided the 'plant competition' category by habitat. We then used another χ^2 test to evaluate whether the contribution of non-native species differed significantly across habitat types—both in relation to other threats and in comparison with expansive native taxa.

3 | RESULTS

Monitoring experts reported 1748 locally relevant threats affecting 1231 populations of 201 target species under Berlin's Flora Protection Programme. The discrepancy between numbers of threats and populations reflects that many populations faced multiple pressures.

3.1 | Major threat factors

Our analysis identified one predominant threat factor (Figure 1, Table 1): Biotic threats accounted for nearly two thirds of all mentions (63.2%), followed by agriculture (27.6%), nutrient influx/pollution (18.2%), urban development (9.2%) and recreational activities (8.2%). Within biotic threats, plant competition was clearly dominant (49.1%), whereas animal-related threats (14.5%) were led by disturbance from wild boar. Agricultural threats were dominated by land abandonment (18.6%), which outweighed intensification (6.8%) and inappropriate use or timing (8.4%). For nutrient influx/pollution, waste disposal and water pollution together accounted for 15.9%, with traffic and other urban emissions contributing 6.8%.

Across all threat categories, a greater proportion of species than populations was affected: on average, 24.8% of species versus 18.5% of populations. The relative importance of threat factors differed significantly between the population and species levels ($\chi^2 = 43.05$, $df = 7$, $p = 3.26 \times 10^{-7}$). For example, forestry-related threats affected only 5% of populations but 12.9% of species.

3.2 | Expansive native and non-native species

Approximately 80% of the biotic threats reported for 778 populations were associated with plant species (Figure 2). Expansive species affected 24.9% of all populations, followed by natural succession (23.2%) and deliberate shrub or tree plantings (11.5%).

The three most frequently mentioned native and non-native species together threaten a quarter of all populations of the target species (see Table S2 for threatened species and population numbers).

Among expansive species, native taxa were reported to pose the greatest threat, affecting 11.5% of populations and 48.3% of target species. The most frequently cited were *Calamagrostis epigejos*, *Phragmites australis* and *Populus tremula*. Other notable natives included various *Rubus* and *Acer* species, *Tanacetum vulgare*, *Pinus*

TABLE 1 Locally relevant threats to 1231 populations of 201 target species (Berlin Flora Protection Programme), assigned by experts based on a biotope mapping scheme (LUB, 2019).

	Threats	Including	Populations affected	%	Species affected	%
1.	**Urban development**	Construction works, road and path construction, filling of stones, etc.	**113**	**9.2**	**54**	**26.9**
2.	**Forestry**	Clearing, clear-cutting, damage caused by felling operations, afforestation, conversion of forest soil	**61**	**5.0**	**26**	**12.9**
3.	**Agriculture**		**340**	**27.6**	**101**	**50.2**
3.1	Intensification of use	Grassland conversion, erosion by wind and water	84	6.8	33	16.4
3.2	Abandonment of uses		229	18.6	68	33.8
3.3	Inappropriate type and period of uses		104	8.4	44	21.9
3.4	Other	Damage from grazing	5	0.4	5	2.5
4.	**Use of water bodies**		**105**	**8.5**	**40**	**19.9**
4.1	Drainage and water withdrawal	Damage due to peat mineralisation	99	8.0	39	19.4
4.2	Other	Water body development, overbanking, mowing of ditches	9	0.7	6	3.0
5.	**Material movement**	Material removal, excavation, backfill	**72**	**5.8**	**36**	**17.9**
6.	**Recreational activities**		**101**	**8.2**	**38**	**18.9**
6.1	Formal recreation and leisure use	Fishing, fireplaces, campsites, jetties	32	2.6	8	4.0
6.2	Informal recreation and leisure use	Damage caused by stepping on, trampling, driving on, bathing, letting in boats	69	5.6	26	12.9
6.3	Unspecified recreation activities		28	2.3	21	10.4
7.	**Biotic threats**		**778**	**63.2**	**157**	**78.1**
7.1	Threats from animals	Damage due to insects, trampling, rooting, browsing	179	14.5	59	29.4
7.2	Competition from plants		605	49.1	133	66.2
7.2.1	Succession	Vegetation inhibiting regeneration	286	23.2	81	40.3
7.2.2	Expansive species		307	24.9	97	48.3
7.2.2.1	Native		141	11.5	71	35.3
7.2.2.2	Non-native		58	4.7	32	15.9
7.2.2.3	Native and non-native		41	3.3	24	11.9
7.2.2.4	Unspecified		67	5.4	24	11.9
7.2.3	Tree and shrub plantings	Deliberately planted taxa and their offspring	142	11.5	58	28.9
7.2.3.1	Non-native		88	7.1	45	22.4
7.3	Other biotic threats		60	4.9	38	18.9
8.	**Nutrient influxes and pollution**		**224**	**18.2**	**85**	**42.3**
8.1	Nutrient influx from agriculture	Pesticides	19	1.5	11	5.5
8.2	Emissions from traffic and urban use	Eutrophication	84	6.8	64	31.8
8.3	Garbage and other material inputs	Garden waste, water pollution	196	15.9	36	17.9
	Total		**1231**		**201**	

Note: The bold highlight indicates the highest threat categories.

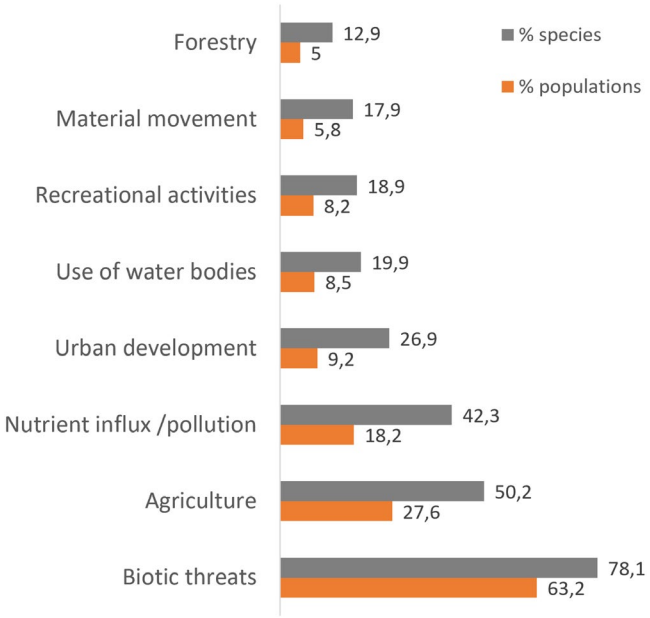


FIGURE 1 Expert-based assignment of principal threat factors to 1231 populations and the proportion of affected target species ($n = 201$) in Berlin's Flora Protection Programme. Multiple threats could be assigned per population. The relative importance of threat factors differs significantly between affected populations and species ($\chi^2 = 43.05$, df = 7, $p = 3.26e{-}07$).

sylvestris, *Urtica dioica*, *Carex* species, *Quercus robur*, *Hedera helix*, several *Salix* species, *Vincetoxicum hirundinaria* and 17 additional taxa.

Non-native expansive species alone threatened 4.5% of populations and 15.9% of the target species. The most frequently cited were *Solidago canadensis* and *S. gigantea*, followed by *Prunus serotina*, *Robinia pseudoacacia*, *Acer negundo* and eight other taxa. Additionally, 41 populations (3.3%)—representing 11.9% of target species—were simultaneously threatened by both native and non-native expansive species. Taking into account the 88 populations threatened by non-native species within the 'Tree and shrub plantings' category, a total of 187 target-species populations (15.2%) are threatened either exclusively or partially by non-native taxa—compared with 197 populations (16.0%) threatened by native species.

3.2.1 | Threats across habitat types

Across all habitat types, threats from non-native species—ranging from 0% to 12.9%—were consistently less often reported than all other threat factors combined (Figure 3). However, the proportion of populations affected by expansive non-native plants varied significantly among habitat types ($\chi^2 = 17.299$, df = 8, $p = 0.0271$). Urban habitats (i.e. habitats within the built environment and on ruderal

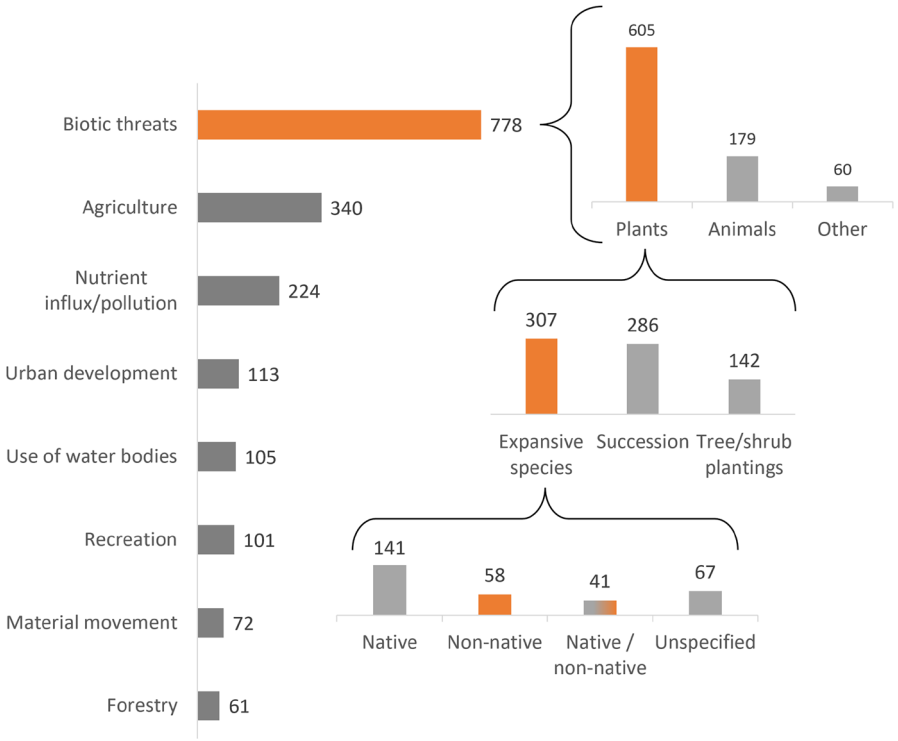


FIGURE 2 Relative importance of different biotic threats to the populations of endangered species, broken down into subcategories to highlight the relevance of expansive non-native species. The sums of subcategories exceed the totals of higher-level categories, as each population was counted only once in each higher level category (see text).

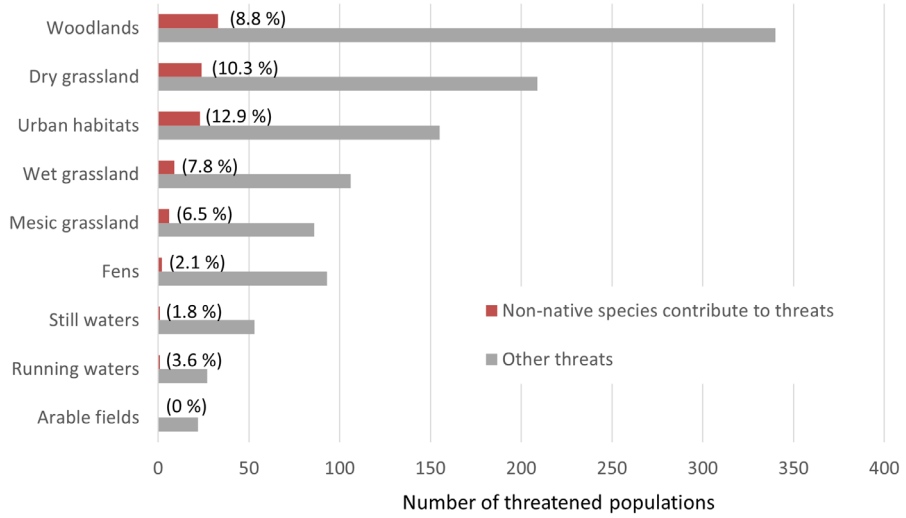


FIGURE 3 Number of target species populations affected by expansive non-native species (alone or in combination with expansive native species; red bars), and number of populations affected by other threats (grey bars). The relative contribution of non-native plant species is indicated in brackets next to the red bars and differs significantly among habitat types ($\chi^2 = 17.299$, df = 8, $p = 0.0271$).

sites) exhibited the highest proportion of populations threatened by non-native plants (12.9%), followed by dry grasslands (10.3%) and woodlands (8.8%). In contrast, populations in arable fields (0%), still waters (1.8%) and fens (2.1%) were least affected.

The relative contribution of expansive non-native species to overall biotic threats, in comparison to native species, did not differ significantly across major habitat types—this applies both to non-native species alone and to their joint impact with native species (Figure 4).

4 | DISCUSSION

Effective urban biodiversity conservation depends on understanding the specific drivers of species decline. While previous reviews have identified a range of threats (de Barros Ruas et al., 2022; Kowarik et al., 2025), they mostly reflect research attention or perceived relevance—rather than providing empirical evidence of threat impacts at the city scale. To address this gap, we quantified (1) the relative importance of diverse threat factors affecting extant populations of endangered plant species in a major European city; (2) the specific role of non-native plant invasions compared to other threats; and (3) how threat patterns vary across major habitat types.

Surprisingly, although non-native species exert pressure on endangered populations, other threats were substantially more important. This advances the discourse beyond isolated invasion case studies in cities, which often lack comprehensive, cumulative impact assessments (Gaertner & Kowarik, 2023). While some comparisons exist for ecosystem services/disservices (Schlaepfer et al., 2020; Tartaglia & Aronson, 2024) or expansive native versus invasive non-native species (Hejda et al., 2021; Kortz et al., 2024; Yazlık & Ambarlı, 2022), the relative importance of native versus non-native species for endangered plants remains largely unknown—yet is essential for setting evidence-based urban conservation priorities.

4.1 | Relative importance of threat factors

A central finding was the dominance of biotic drivers, affecting 63% of populations and 78% of target species. By contrast, agriculture—often the leading threat at larger spatial scales (Hochkirch et al., 2023)—ranks second in Berlin, where abandonment of agricultural practices was a key factor (Table 1). This trend particularly affects low-productivity wet grasslands, which in Berlin often coincide with remnant farmland. In contrast, other grassland types occur more evenly across urban environments (Fischer et al., 2013).

Typical urban pressures—nutrient inputs, pollution, groundwater depletion, recreational use and development—are globally pervasive (de Barros Ruas et al., 2022; Kowarik et al., 2025), but were reported less frequently than biotic threats here. While the monitoring experts possess unparalleled knowledge of the target species' populations and their habitats, they may have over- or underestimated certain factors or their interactions (Hernández-Yáñez et al., 2016). For instance, the effects of habitat fragmentation (Newman et al., 2013; Ramalho et al., 2014), the urban heat island in conjunction with climate change (Solecki & Marcotullio, 2013), or increasing urbanization (Planchuelo et al., 2020a) all matter but may be less observable at the local scale. Likewise, diffuse nutrient inputs are difficult to assess locally (Carey et al., 2013), yet can drive species loss by favouring nitrophilous species, even under consistent site maintenance (Bergamini et al., 2009; Wesche et al., 2012). Experts occasionally reported nutrient influxes from adjacent urban areas, though less frequently than direct garbage deposition.

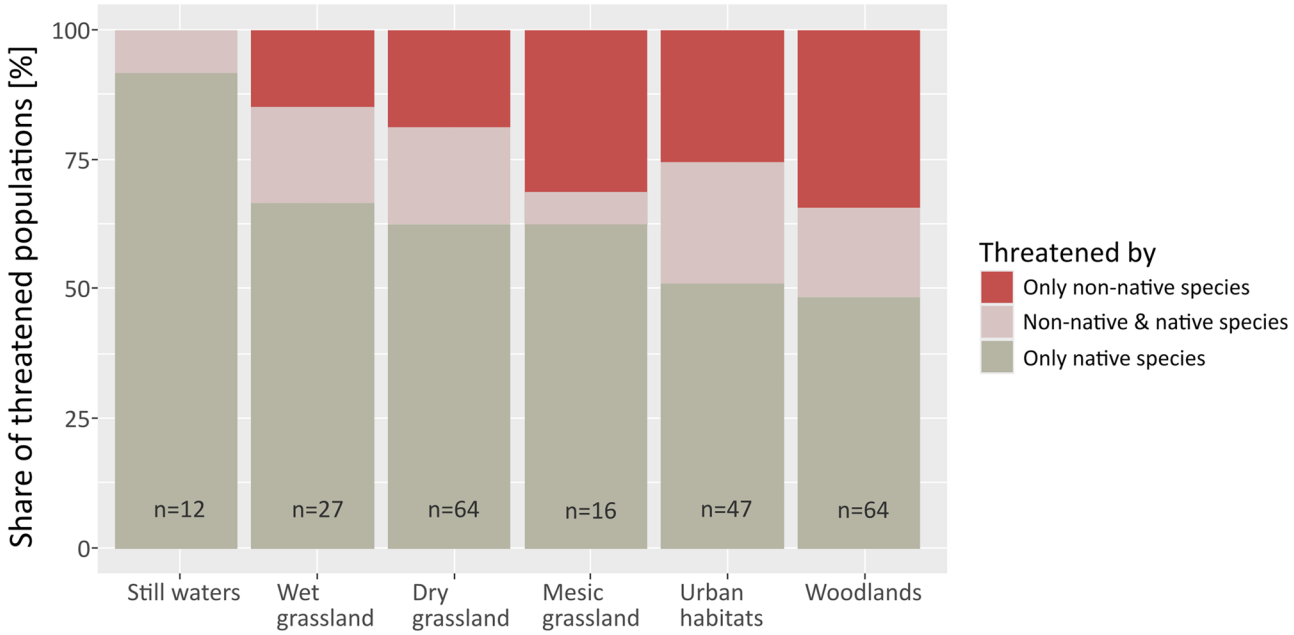


FIGURE 4 Relative contribution of non-native versus native expansive species as biotic threats to populations of target species across six habitat types (n = number of populations threatened by expansive species). Habitat types with fewer than 10 threatened populations are not shown. Neither the contribution of non-native species alone ($\chi^2 = 10.329$, df = 5, $p = 0.06644$) nor the combined contribution of native and non-native species ($\chi^2 = 10.423$, df = 5, $p = 0.06409$) differs significantly across habitat types.

4.2 | Biotic threats

Among biotic threats, plant competition emerged as the primary driver, with expansive species and successional processes exerting nearly equal influence. However, individual threat factors often interact (Hernández-Yáñez et al., 2016); for example, land-use change and increased nutrient inputs in non-urban habitats can trigger both expansion of dominant species and succession, displacing habitat specialists with more competitive taxa (Harásek et al., 2023; Hilpold et al., 2018). In Australian urban reserves, fire suppression represents an additional driver of ecological change (Zeeman et al., 2017). Similarly, the spread of expansive species and progressing succession in Berlin likely reflect broader environmental shifts. In other words, expansive species—whether native or non-native—are often not the drivers of change, but rather passengers in the process (Bauer, 2012).

4.2.1 | Native and non-native species

Among expansive species, native taxa were mentioned more often than non-natives, affecting 11.5% of populations compared to 4.5%, with an additional 3.3% threatened by both. The experts cited 'tree and shrub plantings' as another threat factor, including both non-native (e.g. *R. pseudoacacia*, *P. serotina* and *A. negundo*) and native species (e.g. *Acer* taxa, *Fagus sylvatica*). The threats posed by these species cannot always be attributed to invasion processes, as no systematic distinction was made between plantings and their offspring. When all mentions of non-native taxa are taken into account, the proportion of populations threatened by these species rises to 15.2%, which is slightly lower than the 16.0% of populations threatened by native taxa. Consequently, non-native species play a similar role to native species—and a clearly minor role compared with other threat factors.

This relatively minor role contrasts with the commonly assumed high importance of plant invasions as a driver of species decline in cities (de Barros Ruas et al., 2022). This is particularly striking in Berlin—a known invasion hotspot where established neophytes make up 20.1% of the flora (Seitz et al., 2018) and up to 32% in certain vegetation types (Kowarik, 2008). While targeted control of non-native species is useful (Potgieter et al., 2022), our findings suggest that non-native plants do not necessarily translate into the most pressing conservation conflicts in cities.

The most common expansive native and non-native species share key characteristics: they spread clonally and are favoured by land abandonment and nutrient enrichment. The native *Calamagrostis epigejos* rapidly out-competes other species by forming dense clonal populations; nitrogen deposition and reduced mowing promote its spread in grasslands and pine plantations, as well as natural succession on vacant land (Hejda et al., 2021; Rebele & Lehmann, 2001, 2016). Consistently, many affected target species are typically associated with low-nutrient habitats (Table S2), such as the fern *Botrychium lunaria* and the sedges *Carex hartmaniorum* and *C. supina*, all of which are threatened with extinction in Berlin (Seitz et al., 2018). The native *Phragmites australis* threatens wetland specialists in fens and wet grasslands, particularly under conditions of increased eutrophication and lacking mowing (Güsewell, 2003; Packer et al., 2017),

including *Rhynchospora alba*, *Carex viridula* and several *Sphagnum* species (Table S2). Ironically, the main Berlin habitat of *P. australis* (reed beds) has long been in decline (Sukopp & Markstein, 1989). The native tree *Populus tremula* mostly invades dry grasslands by clonal growth and threatens open habitat specialists (e.g. *Koeleria glauca*, *Pulsatilla pratensis* subsp. *nigricans*; Table S2).

Clonal stands of non-native *Solidago* species suppress other plants (Weber, 2000; Weber & Jakobs, 2005), particularly under nutrient-enriched conditions (Ren et al., 2022) and following land-use abandonment (Świerszcz et al., 2024). In this study, *Solidago* taxa threaten species of both dry grassland (e.g. *Carex hartmaniorum*, *Silene tatarica*) and wet grassland (e.g. *Carex viridula*, *Gentiana pneumonanthe*; Table S2). The non-native tree *Prunus serotina*, widely planted in Berlin's forests, can dominate near former planting sites and spread into drained moorlands, forest clearings, hedgerows and heaths (Deckers et al., 2005; Starfinger et al., 2003). In Berlin, it consequently shades out endangered species of open forests (e.g. *Botrychium matricariifolium*), wet habitats (e.g. *Osmunda regalis*) and dry grassland (e.g. *Scorzonera purpurea*). Finally, the tree *Robinia pseudoacacia* also benefits from reduced land use (Cierjacks et al., 2013), replacing similar dry grassland species mentioned above (Table S2).

Strikingly, none of the EU-listed Invasive Alien Species of Union Concern—despite being common in Berlin (e.g. *Ailanthus altissima*, *Impatiens glandulifera* and *Heracleum mantegazzianum*; Seitz et al., 2012)—were reported as threats by local experts. This suggests that the mere presence of invasive species does not automatically endanger conservation targets, likely because they occupy different habitats. Nonetheless, invasive species may occur in urban nature reserves and should be monitored for potential competition with conservation target species—and managed if necessary (Jarošik et al., 2011; Nelufule et al., 2024).

While our findings highlight the relatively minor role of non-native species as a threat to endangered plant populations in Berlin, they should not be overgeneralized. Several caveats must be considered when interpreting the comparison between native and non-native species.

First, comparing the frequency with which native versus non-native species are reported as threats does not capture potential differences in the *intensity* or *long-term persistence* of their impacts. For example, the native pioneer tree *Betula pendula* and the introduced *Robinia pseudoacacia* both shade target species of open habitats. However, the impacts of *R. pseudoacacia* are more intense and persistent due to its nitrogen fixation and vigorous clonal growth (Cierjacks et al., 2013). Whether such functional differences systematically distinguish native from non-native expansive species remains an open question.

Second, our study does not allow conclusions about the extent to which native and non-native expansive species act as drivers or merely as passengers of environmental change (Bauer, 2012). The examples discussed above suggest that land-use change and nutrient enrichment may be the underlying mechanisms facilitating the success of both native and non-native expansive species. Consistently, many target species—typically associated with low-nutrient habitats—are affected by both native and non-native expansive species (Table S2), which, in turn, benefit from nutrient inputs and land-use change (as discussed above). Examples include the fern *Botrychium lunaria*, the grass *Koeleria glauca* and the sedge species *Carex hartmaniorum* and *C. supina*. However, two key questions remain open for future investigations: whether native and non-native expansive species depend to different extents on habitat changes as a prerequisite for their spread, and whether there are functional differences between the target species affected.

A third caveat is that invasion impacts depend on the biogeographical context. In cities with different climates or those adjacent to highly invasible biomes—such as the South African fynbos (Potgieter et al., 2020)—plant invasions may pose much greater risks than in Central European cities (Gaertner & Kowarik, 2023). Finally, future threats cannot be ruled out, and even seemingly, minor pressures may jeopardize the last remaining populations of endangered species. However, this caveat applies to all types of threat factors.

4.2.2 | Non-native species across habitat types

Expansive non-native species were consistently less important than other individual threat factors at the habitat level. However, their relative importance varied substantially among habitat types (Figure 3). High proportions of non-native threats in urban habitats and woodlands, and low proportions in aquatic and fen ecosystems, align with broader Central European patterns of habitat invasibility (Chytrý et al., 2005, 2008). Although European grasslands generally show a low invasion level (Axmanová et al., 2021), dry grasslands in Berlin had the second-highest share of non-native threats. This is likely due to their proximity to urban propagule sources, as they are often embedded in densely urbanized areas (von der Lippe et al., 2020). In contrast, wet grasslands typically occur in agricultural landscapes and are therefore less exposed to urban invasion pressure (Fischer et al., 2013).

When considering all biotic threats, native expansive species were overall cited more frequently than non-native species. However, their relative proportions did not differ significantly (Figure 4). It should be noted that the relatively broad classification into six habitat types may obscure finer scale variation in the impacts of non-native species—differences that might become apparent with more detailed habitat differentiation.

4.3 | Succession

Succession posed a threat comparable in magnitude to that of expansive species. On agricultural land, it is primarily driven by abandonment; in woodlands and wastelands, by the promotion of natural forest dynamics. As open and grassland stages undergo shrub encroachment and forest development, vacant land experiences rapid species turnover (Rebele & Lehmann, 2016; Westermann

et al., 2011). However, trends in species richness during succession are scale-dependent (Staude et al., 2023): while overall species richness may increase at larger scales (Salisbury et al., 2021), it often declines in small plots when clonally growing herbs, grasses and woody species out-compete less competitive target species—mostly herbs and geophytes (Planchuelo et al., 2020a).

Succession also plays an important role in Berlin's forests, which support the largest number of target species populations (Planchuelo et al., 2019). Light pine forests, pine-oak woodlands, sparse forest edges and clearings—once common due to historical forest management—are becoming denser or overgrown, disadvantaging species adapted to semi-open habitats (Seidling, 1990). Moreover, deliberate tree plantings and their offspring (both native and non-native) threaten 11.5% of the populations. Current efforts to implement more natural forest management—such as promoting multi-layered hardwood stands with dense canopy cover—may conflict with the conservation needs of highly endangered species, including *Botrychium multifidum*, *B. matricariifolium* and *Chimaphila umbellata*.

To reconcile the objectives of developing near-natural forests and conserving species adapted to semi-open habitats, it is essential to preserve open areas within forest stands—for example, by grazing, mowing or sapling removal. For instance, the removal of *Acer platanoides* and *Prunus serotina* from the shrub layer has supported populations of the highly endangered *Scorzonera humilis* and *Potentilla alba* in a historically grazed woodland (Meißner & Rohner, 2020).

4.4 | Limitations

This study focuses on current threats to extant populations of endangered plant species and cannot address the drivers of past extinctions in populations that no longer exist. As with other expert-based assessments conducted at supra-regional (Chamberlain et al., 2016), national (Hernández-Yáñez et al., 2016), or continental scales (Burgman et al., 2007; Hochkirch et al., 2023), our approach carries inherent uncertainties: experts may over- or underestimate the effects of individual or interacting threats, particularly those related to complex urban environments (see Section 4.1). However, given the large sample size of site-level assessments, we argue that the overall identification and ranking of threat factors is sufficiently robust to support conservation planning and prioritization (cf. Schwartz et al., 2018).

5 | CONCLUSIONS

Urban environments can support diverse habitats for endangered plant species, yet local populations remain subject to substantial extinction debts driven by multiple, site-specific threats—even within designated protected areas. Drawing on expert assessments of 1231 populations representing 201 target species in Berlin, we identified the relative importance of these threats to inform conservation priorities and guide targeted management.

Our results show that biotic pressures—particularly competition from expansive and successional species—constitute the most pervasive threats. This underscores the limitations of passive habitat protection, which fails to address dynamic processes such as vegetation encroachment and nutrient enrichment. As these drivers often stem from land-use change, interventions that target their underlying causes—for example, through reinstating traditional grazing or mowing regimes and reducing nutrient inputs—are likely to be more effective than narrowly focused, species-specific measures. Ensuring habitat quality at the local scale thus emerges as a central challenge for urban plant conservation. Given the variation in threat profiles among habitat types, coordinated action across sectors—including conservation, forestry, agriculture and urban land management—is essential.

Although non-native species are widely perceived as key threats to urban biodiversity, our data suggest they impact only 15.2% of endangered plant populations—a similar share compared to threats by native species (16.0%). This implies that public and expert concern may overemphasize their role while underestimating more significant drivers of decline. Robust, context-specific risk assessments are therefore critical to realign conservation priorities with actual threat levels.

We conclude that evidence-based, comparative threat assessments—across cities with contrasting biogeographic and socio-economic settings—are key to identifying the most urgent and manageable conservation challenges in urban landscapes. Such approaches are essential to optimize the allocation of resources and improve outcomes for endangered plant species in cities.

AUTHOR CONTRIBUTIONS

Ingo Kowarik and Moritz von der Lippe conceived the ideas and designed methodology; Justus Meißner provided the data and supporting information; Moritz von der Lippe and Valén Holle processed and analysed the data; Ingo Kowarik led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors state no conflict of interest.

DATA AVAILABILITY STATEMENT

Data are available from the Dryad Digital Repository https://doi.org/10.5061/dryad.bzkh189qv (von der Lippe et al., 2026).

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REFERENCES

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M., … Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, *281*(1780), 20133330.

Axmanová, I., Kalusová, V., Danihelka, J., Dengler, J., Pergl, J., Pyšek, P., Večeřa, M., Attorre, F., Biurrun, I., Boch, S., Conradi, T., Gavilán, R. G., Jiménez-Alfaro, B., Knollová, I., Kuzemko, A., Lenoir, J., Leostrin, A., Medvecká, J., Moeslund, J. E., … Chytrý, M. (2021). Neophyte invasions in European grasslands. *Journal of Vegetation Science*, *32*, e12994.

Bauer, J. T. (2012). Invasive species: "Back-seat drivers" of ecosystem change? *Biological Invasions*, *14*, 1295–1304.

Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters*, *18*(6), 581–592.

Bergamini, A., Peintinger, M., Fakheran, S., Moradi, H., Schmid, B., & Joshi, J. (2009). Loss of habitat specialists despite conservation management in fen remnants 1995–2006. *Perspectives in Plant Ecology, Evolution and Systematics*, *11*(1), 65–79.

Burgman, M. A., Keith, D., Hopper, S. D., Widyatmoko, D., & Drill, C. (2007). Threat syndromes and conservation of the Australian flora. *Biological Conservation*, *134*(1), 73–82.

Carey, R. O., Hochmuth, G. J., Martinez, C. J., Boyer, T. H., Dukes, M. D., Toor, G. S., & Cisar, J. L. (2013). Evaluating nutrient impacts in urban watersheds: Challenges and research opportunities. *Environmental Pollution*, *173*, 138–149.

Catford, J. A., Wilson, J. R. U., Pyšek, P., Hulme, P. E., & Duncan, R. P. (2022). Addressing context dependence in ecology. *Trends in Ecology & Evolution*, *37*, 158–170.

CBD. (2022). *Kunming-Montreal global biodiversity framework*. *CBD/COP/DEC/15/4*. Convention on Biological Diversity.

Chamberlain, D. E., Pedrini, P., Brambilla, M., Rolando, A., & Girardello, M. (2016). Identifying key conservation threats to Alpine birds through expert knowledge. *PeerJ*, *4*, e1723.

Chocholoušková, Z., & Pyšek, P. (2003). Changes in composition and structure of urban flora over 120 years: A case study of the city of Plzeň. *Flora*, *198*(5), 366–376.

Chytrý, M., Jarosík, V., Pyšek, P., Hájek, O., Knollová, I., Tichý, L., & Danihelka, J. (2008). Separating habitat invasibility by alien plants from the actual level of invasion. *Ecology*, *89*, 1541–1553.

Chytrý, M., Pyšek, P., Tichý, L., Knollová, I., & Danihelka, J. (2005). Invasions by alien plants in the Czech Republic: A quantitative assessment across habitats. *Preslia*, *77*, 339–354.

Cierjacks, A., Kowarik, I., Joshi, J., Hempel, S., Ristow, M., von der Lippe, M., & Weber, E. (2013). Biological flora of the British Isles: *Robinia pseudoacacia*. *Journal of Ecology*, *101*(6), 1623–1640.

de Barros Ruas, R., Costa, L. M. S., & Bered, F. (2022). Urbanization driving changes in plant species and communities–A global view. *Global Ecology and Conservation*, *38*, e02243.

Deckers, B., Verheyen, K., Hermy, M., & Muys, B. (2005). Effects of landscape structure on the invasive spread of black cherry *Prunus serotina* in an agricultural landscape in Flanders, Belgium. *Ecography*, *28*(1), 99–109.

Dehnen-Schmutz, K., Pescott, O. L., Booy, O., & Walker, K. J. (2022). Integrating expert knowledge at regional and national scales improves impact assessments of non-native species. *NeoBiota*, *77*, 79–100.

Drescher, M., Perera, A. H., Johnson, C. J., Buse, L. J., Drew, C. A., & Burgman, M. A. (2013). Toward rigorous use of expert knowledge in ecological research. *Ecosphere*, *4*(7), 1–26.

Duncan, R. P., Clemants, S. E., Corlett, R. T., Hahs, A. K., McCarthy, M. A., McDonnell, M. J., Schwartz, M. W., Thompson, K., Vesk, P. A., & Williams, N. S. G. (2011). Plant traits and extinction in urban areas: A meta-analysis of 11 cities. *Global Ecology and Biogeography*, *20*(4), 509–519.

Fischer, L. K., von der Lippe, M., & Kowarik, I. (2013). Urban land use types contribute to grassland conservation: The example of Berlin. *Urban Forestry & Urban Greening*, *12*(3), 263–272.

Gaertner, M., & Kowarik, I. (2023). Plant invasions in cities: Patterns, processes, consequences, and management. In *Routledge handbook of urban biodiversity* (pp. 172–190). Routledge.

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

Güsewell, S. (2003). Management of *Phragmites australis* in Swiss fen meadows by mowing in early summer. *Wetlands Ecology and Management*, *11*(6), 433–445.

Hahs, A. K., McDonnell, M. J., McCarthy, M. A., Vesk, P. A., Corlett, R. T., Norton, B. A., Clemants, S. E., Duncan, R. P., Thompson, K., Schwartz, M. W., & Williams, N. S. G. (2009). A global synthesis of plant extinction rates in urban areas. *Ecology Letters*, *12*(11), 1165–1173.

Harásek, M., Klinkovská, K., & Chytrý, M. (2023). Vegetation change in acidic dry grasslands in Moravia (Czech Republic) over three decades: Slow decrease in habitat quality after grazing cessation. *Applied Vegetation Science*, *26*(2), e12726.

Hejda, M., Sádlo, J., Kutlvašr, J., Petřík, P., Vítková, M., Vojík, M., Pyšek, P., & Pergl, J. (2021). Impact of invasive and native dominants on species richness and diversity of plant communities. *Preslia*, *93*(3), 181–201.

Hernández-Yáñez, H., Kos, J. T., Bast, M. D., Griggs, J. L., Hage, P. A., Killian, A., Loza, M. I., Whitmore, M. B., & Smith, A. B. (2016). A systematic assessment of threats affecting the rare plants of the United States. *Biological Conservation*, *203*, 260–267.

Hilpold, A., Seeber, J., Fontana, V., Niedrist, G., Rief, A., Steinwandter, M., Tasser, E., & Tappeiner, U. (2018). Decline of rare and specialist species across multiple taxonomic groups after grassland intensification and abandonment. *Biodiversity and Conservation*, *27*(14), 3729–3744.

Hochkirch, A., Bilz, M., Ferreira, C. C., Danielczak, A., Allen, D., Nieto, A., Rondinini, C., Harding, K., Hilton-Taylor, C., Pollock, C. M., Seddon, M., Vié, J.-C., Alexander, K. N. A., Beech, E., Biscoito, M., Braud, Y., Burfield, I. J., Buzzetti, F. M., Cálix, M., … Zuna-Kratky, T. (2023). A multi-taxon analysis of European Red Lists reveals major threats to biodiversity. *PLoS One*, *18*(11), e0293083.

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*(1), 117–126.

Jarošik, V., Pyšek, P., & Kadlec, T. (2011). Alien plants in urban nature reserves: From red-list species to future invaders? *NeoBiota*, *10*, 27–46.

Knapp, S., von der Lippe, M., & Kowarik, I. (2022). Interactions of functional traits with native status and ecosystem novelty explain the establishment of plant species within urban ecosystems: Evidence from Berlin, Germany. *Frontiers in Ecology and Evolution*, *10*, 790340.

Kortz, A., Hejda, M., Pergl, J., Kutlvašr, J., Petřík, P., Sádlo, J., Vítková, M., Vojík, M., & Pyšek, P. (2024). Impacts of native and alien plant dominants at different spatial scales. *NeoBiota*, *92*, 29–43.

Kowarik, I. (2008). On the role of alien species in urban flora and vegetation. In *Urban ecology: An international perspective on the interaction between humans and nature* (pp. 321–338). Springer.

Kowarik, I. (2023). Urban biodiversity, ecosystems and the city. Insights from 50 years of the Berlin school of urban ecology. *Landscape and Urban Planning*, *240*, 104877.

Kowarik, I., Fischer, L. K., Haase, D., Kabisch, N., Kleinschroth, F., Konijnendijk, C., Straka, T. M., & von Haaren, C. (2025). Promoting

urban biodiversity for the benefit of people and nature. *Nature Reviews Biodiversity*, *1*, 214–232.
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.
Krausch, H. D., & Sukopp, H. (2010). Die Geschichte der geobotanischen Forschung in Berlin und Brandenburg. *Verhandlungen des Botanischen Vereins Berlin-Brandenburg, Beiheft*, *6*, 5–155.
Kühn, I., Brandl, R., & Klotz, S. (2004). The flora of German cities is naturally species rich. *Evolutionary Ecology Research*, *6*(5), 749–764.
Lepczyk, C. A., Aronson, M. F. J., & La Sorte, F. A. (2023). Cities as sanctuaries. *Frontiers in Ecology and the Environment*, *21*(5), 251–259.
LUB. (2019). Biotopkartierung Brandenburg, Band 1, Kartierungsanleitung [Landesumweltamt Brandenburg, ed.].
Martin, T. G., Burgman, M. A., Fidler, F., Kuhnert, P. M., Low-Choy, S., McBride, M., & Mengersen, K. (2012). Eliciting expert knowledge in conservation science. *Conservation Biology*, *26*(1), 29–38.
Meißner, J., & Rohner, M. S. (2020). Bericht über die Exkursion anlässlich der 80. Geburtstage von Jürgen Klawitter und Gunther Klemm durch die Wuhlheide am 16. August 2020. *Verhandlungen des Botanischen Vereins Berlin-Brandenburg*, *152*, 275–280.
Meißner, J., & Seitz, B. (2010). Das Florenschutzkonzept im Land Berlin und seine Umsetzung. *Verhandlungen des Botanischen Vereins von Berlin und Brandenburg*, *143*, 111–120.
Nelufule, T., Thenga, T. C., Shivambu, T. C., Shivambu, N., Moshobane, M. C., Seoraj-Pillai, N., & Nangammbi, T. C. (2024). Alien plant species richness in urban protected biodiversity areas: A case study of Tshwane metropolitan municipality, South Africa. *Diversity*, *16*(8), 461.
Newman, B. J., Ladd, P., Brundrett, M., & Dixon, K. W. (2013). Effects of habitat fragmentation on plant reproductive success and population viability at the landscape and habitat scale. *Biological Conservation*, *159*, 16–23.
Packer, J. G., Meyerson, L. A., Skálová, H., Pyšek, P., & Kueffer, C. (2017). Biological flora of the British Isles: *Phragmites australis*. *Journal of Ecology*, *105*(4), 1123–1162.
Piana, M. R., Aronson, M. F. J., Pickett, S. T., & Handel, S. N. (2019). Plants in the city: Understanding recruitment dynamics in urban landscapes. *Frontiers in Ecology and the Environment*, *17*(8), 455–463.
Planchuelo, G., Kowarik, I., & von der Lippe, M. (2020a). Plant traits, biotopes and urbanization dynamics explain the survival of endangered urban plant populations. *Journal of Applied Ecology*, *57*(8), 1581–1592.
Planchuelo, G., Kowarik, I., & von der Lippe, M. (2020b). Endangered plants in novel urban ecosystems are filtered by strategy type and dispersal syndrome, not by spatial dependence on natural remnants. *Frontiers in Ecology and Evolution*, *8*, 18.
Planchuelo, G., von der Lippe, M., & Kowarik, I. (2019). Untangling the role of urban ecosystems as habitats for endangered plant species. *Landscape and Urban Planning*, *189*, 320–334.
Potgieter, L. J., Douwes, E., Gaertner, M., Measey, J., Paap, T., & Richardson, D. M. (2020). Biological invasions in South Africa's urban ecosystems: Patterns, processes, impacts, and management. *Biological Invasions in South Africa*, *14*, 275.
Potgieter, L. J., Gaertner, M., Kueffer, C., Larson, B. M., Livingstone, S. W., O'Farrell, P., & Richardson, D. M. (2017). Alien plants as mediators of ecosystem services and disservices in urban systems: A global review. *Biological Invasions*, *19*, 3571–3588.
Potgieter, L. J., Shrestha, N., & Cadotte, M. W. (2022). Prioritizing terrestrial invasive alien plant species for management in urban ecosystems. *Journal of Applied Ecology*, *59*(3), 872–883.
Ramalho, C. E., Laliberté, E., Poot, P., & Hobbs, R. J. (2014). Complex effects of fragmentation on remnant woodland plant communities of a rapidly urbanizing biodiversity hotspot. *Ecology*, *95*(9), 2466–2478.
Rastandeh, A., Pedersen Zari, M., & Brown, D. K. (2018). Components of landscape pattern and urban biodiversity in an era of climate change: A global survey of expert knowledge. *Urban Ecosystems*, *21*(5), 903–920.
Rebele, F., & Lehmann, C. (2001). Biological flora of central Europe: *Calamagrostis epigejos* (L.) Roth. *Flora*, *196*(5), 325–344.
Rebele, F., & Lehmann, C. (2016). Twenty years of woodland establishment through natural succession on a sandy landfill site in Berlin, Germany. *Urban Forestry & Urban Greening*, *18*, 182–189.
Ren, G., Yang, B., Cui, M., Dai, Z., Xiang, Y., Zhang, H., Li, G., Li, J., Javed, Q., & Du, D. (2022). Warming and elevated nitrogen deposition accelerate the invasion process of *Solidago canadensis* L. *Ecological Processes*, *11*(1), 62.
Salisbury, A., Gallagher, F. J., Parag, H. A., Meneses-Florián, L., & Holzapfel, C. (2021). Plant diversity increases in an urban wildland after four decades of unaided vegetation development in a post-industrial site. *Urban Ecosystems*, *24*(1), 95–111.
Sax, D. F., Schlaepfer, M. A., & Olden, J. D. (2022). Valuing the contributions of non-native species to people and nature. *Trends in Ecology & Evolution*, *37*(12), 1058–1066.
Schlaepfer, M. A., Guinaudeau, B. P., Martin, P., & Wyler, N. (2020). Quantifying the contributions of native and non-native trees to a city's biodiversity and ecosystem services. *Urban Forestry & Urban Greening*, *56*, 126861.
Schwartz, M. W., Cook, C. N., Pressey, R. L., Pullin, A. S., Runge, M. C., Salafsky, N., Sutherland, W. J., & Williamson, M. A. (2018). Decision support frameworks and tools for conservation. *Conservation Letters*, *11*(2), e12385.
Seidling, W. (1990). Recent changes in forest vegetation in an area on the outskirts of Berlin. In H. Sukopp, S. Hejny, & I. Kowarik (Eds.), *Plants and plant communities in the urban environment* (p. 223). SPB Academic Publishing.
Seitz, B., Ristow, M., Meißner, J., Machatzi, B., & Sukopp, H. (2018). Rote Liste und Gesamtartenliste der etablierten Farn- und Blütenpflanzen von Berlin. In: *Der Landesbeauftragte für Naturschutz und Landschaftspflege*. Senatsverwaltung für Umwelt Klima und Verkehr (Ed.) Rote Listen der gefährdeten Pflanzen, Pilze und Tiere von Berlin https://doi.org/10.14279/depositonce-6689
Seitz, B., Ristow, M., Prasse, R., Machatzi, B., Klemm, G., Böcker, R., & Sukopp, H. (2012). Der Berliner Florenatlas. *Verhandlungen Botanischer Verein Berlin Brandenburg, Sonderband*, *7*, 1–533.
Shwartz, A., Turbé, A., Julliard, R., Simon, L., & Prévot, A. C. (2014). Outstanding challenges for urban conservation research and action. *Global Environmental Change*, *28*, 39–49.
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), e2117297119.
Soanes, K., & Lentini, P. E. (2019). When cities are the last chance for saving species. *Frontiers in Ecology and the Environment*, *17*(4), 225–231.
Solecki, W., & Marcotullio, P. J. (2013). Climate change and urban biodiversity vulnerability. In *Urbanization, biodiversity and ecosystem services: Challenges and opportunities: A global assessment* (pp. 485–504). Springer.
Starfinger, U., Kowarik, I., Rode, M., & Schepker, H. (2003). From desirable ornamental plant to pest to accepted addition to the flora?–the perception of an alien tree species through the centuries. *Biological Invasions*, *5*, 323–335.
Staude, I. R., Weigelt, A., & Wirth, C. (2023). Biodiversity change in light of succession theory. *Oikos*, *2023*(11), e09883.
Sukopp, H., & Markstein, B. (1989). Changes of the reed beds along the Berlin Havel, 1962–1987. *Aquatic Botany*, *35*(1), 27–39.
Świerszcz, S., Czarniecka-Wiera, M., Szymura, T. H., & Szymura, M. (2024). From invasive species stand to species-rich grassland: Long-term changes in plant species composition during *Solidago* invaded site restoration. *Journal of Environmental Management*, *353*, 120216.

Tartaglia, E. S., & Aronson, M. F. J. (2024). Plant native: Comparing biodiversity benefits, ecosystem services provisioning, and plant performance of native and non-native plants in urban horticulture. *Urban Ecosystems*, *27*, 2587–2611.

von der Lippe, M., Buchholz, S., Hiller, A., Seitz, B., & Kowarik, I. (2020). CityScapeLab Berlin: A research platform for untangling urbanization effects on biodiversity. *Sustainability*, *12*(6), 2565.

von der Lippe, M., Kowarik, I., Holle, V., & Meißner, J. (2026). Data from: Relative impact of native and non-native plants on endangered species in a major European city. *Dryad Digital Repository*. https://doi.org/10.5061/dryad.bzkh189qv

Weber, E. (2000). Biological flora of central Europe: *Solidago altissima* L. *Flora*, *195*(2), 123–134.

Weber, E., & Jakobs, G. (2005). Biological flora of central Europe: *Solidago gigantea* Aiton. *Flora*, *200*(2), 109–118.

Wesche, K., Krause, B., Culmsee, H., & Leuschner, C. (2012). Fifty years of change in Central European grassland vegetation: Large losses in species richness and animal-pollinated plants. *Biological Conservation*, *150*(1), 76–85.

Westermann, J. R., von der Lippe, M., & Kowarik, I. (2011). Seed traits, landscape and environmental parameters as predictors of species occurrence in fragmented urban railway habitats. *Basic and Applied Ecology*, *12*(1), 29–37.

Williams, N. S., Hahs, A. K., & Vesk, P. A. (2015). Urbanisation, plant traits and the composition of urban floras. *Perspectives in Plant Ecology, Evolution and Systematics*, *17*(1), 78–86.

Yazlık, A., & Ambarlı, D. (2022). Do non-native and dominant native species carry a similar risk of invasiveness? A case study for plants in Turkey. *NeoBiota*, *76*, 53–72.

Zeeman, B. J., McDonnell, M. J., Kendal, D., & Morgan, J. W. (2017). Biotic homogenization in an increasingly urbanized temperate grassland ecosystem. *Journal of Vegetation Science*, *28*(3), 550–561.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Table S1. Target species of the Berlin Flora Protection Program and their population numbers across major habitat types.

Table S2. The three most frequently mentioned expansive native and non-native species as threats to the populations of the target species in the Berlin Flora Protection Program. The number behind the species name refers to the number of affected populations of the target species.

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