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REVIEW PAPER

Biodiversity in residential gardens: a review of the evidence base

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Abstract
Residential gardens are a principal component of urban green infrastructure throughout the world and their potential positive contributions to biodiversity are increasingly recognised. But the characteristics of gardens reflect the needs, values and interests of individual households. The present review summarises evidence from studies of garden biodiversity published in the scientific literature, describes major themes and identifies important knowledge gaps. A search of the Web of Science database identified 408 published articles on the biodiversity of residential gardens (1981–2022), with numbers increasing over time and a strong bias towards Europe (32.1%) and North America (23.8%). Plants and invertebrates were most frequently studied, and species diversity was often correlated with garden size and habitat complexity. Botanic composition and vegetation cover were often positively associated with the diversity and abundance of fauna. Non-native plants contributed substantially to garden plant diversity and evidence from some studies indicated benefits to other species linked to their functional attributes. Intensive management including frequent lawn mowing, fertiliser and pesticide application, and a more formal, 'neater' garden appearance were often associated with reduced biodiversity. However, results varied amongst studies, for example in relation to the impacts of mowing frequency on lawn diversity. There was a general paucity of experimental evidence on the impacts of different management regimes on garden biodiversity and few replicated experimental tests of recommended 'wildlife-friendly practices'. Several studies identified the importance of connectivity amongst gardens and with other green infra-structure for species dispersal and ecosystem functioning. Emerging threats to garden biodiversity include their replacement by development, conversion to hard surfaces and declining plot sizes. Managing these challenges and maximising the biodiversity value of residential gardens requires greater engagement from policymakers and planners, and partnerships between public bodies and private households to co-ordinate local initiatives.

Keywords Green infrastructure · Backyards · Species richness · Wildlife gardening

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Introduction

The domestic or residential garden, also known in parts of the world as a 'backyard', may be described as an enclosed area of land associated with a domestic dwelling and usually devoted (at least in part) to a lawn, flowers, trees, fruits, vegetables and/or other useful plants. Small scale food production on plots adjacent to human settlements is perhaps the oldest and most enduring form of cultivation (Niñez 1987). The practice of gardening dates back to ancient times when humans first began sedentary cultivation, with the earliest records of gardens originating from the Middle-East (Campbell 2019). Today, gardens are widely associated with residential dwellings throughout the world and their design reflects their purpose which varies from the highly utilitarian to the purely aesthetic. At the utilitarian end of this spectrum are so-called 'home gardens' which can be described as small-scale agroforestry systems adjacent to residential dwellings and traditionally associated with subsistence in low-income communities (Niñez 1987). At the other extreme are the highly designed modern ornamental gardens that typify affluent residential neighbourhoods. However, most private residential gardens are likely to be intermediate in form, blending ornamental planting with small scale food production, recreational space and parking areas for vehicles, in varying degrees. What all gardens have in common however is that they can be regarded as 'designer ecosystems' (Light et al. 2013) with characteristics that reflect the local environment and the needs, values and interests of individual households.

Domestic gardens may not only be a source of sustenance and raw materials but are also areas for recreation and relaxation. The public health benefits of gardens have been well described, in terms of improved mental health and well-being, increased physical activity and a source of healthy homegrown nutrition (Chalmin-Pui et al. 2021; de Bell et al. 2020; Soga et al. 2017). Gardens have also been identified as an important force for social cohesion (Schram-Bijkerk et al. 2018). In addition, they may play a pivotal role in developing positive attitudes towards nature conservation (Cosquer et al. 2012; Evans et al. 2005), not least because for many people they constitute their main source of contact with the natural world (Bhatti and Church 2001; Dunnett and Qasim 2000), including in the form of views from windows (Cox et al. 2017).

Inevitably, private domestic gardens are concentrated in more densely populated areas. It has been estimated that urban areas may account for up to 0.69% of global land cover (Zhao et al. 2022) and cities are home to about 55% of the human population with the expectation that this will increase to 60% by 2030 (United Nations 2018). Estimates of the total urban area covered by private gardens in Europe and New Zealand range from 16 to 36% (Goddard et al. 2010), with gardens accounting for 35–47% of all urban green space in two UK cities (Loram et al. 2007). Domestic gardens are therefore a principal component of the green infrastructure of the built environment in parts of the world and are recognised as an important source of biodiversity (e.g. Goddard et al. 2010; Loram et al. 2007). Furthermore, their relative importance as sources of biodiversity is likely to increase with the spread of urbanisation, although where pressure on living space is intense then gardens may be squeezed out. Increasing recognition that gardens can make a significant positive contribution to biodiversity in urban environments and the delivery of ecosystem services, is embodied in the United Nations Sustainable Development Goals and in other global policy initiatives that seek to place biodiversity concerns at the heart of urban planning and design (Beumer 2018). The extent of private domestic gardens outside urban areas is however less well described, although there is evidence that these too may provide important ecosystem services to adjacent agricultural land (Samnegard et al. 2011).

In recent decades there has been an increase in media coverage, and public interest in the potential biodiversity benefits of gardens, fostered by conservation groups and some gardening organisations that have popularised the concept of 'wildlife-friendly' gardening'. This has been accompanied by a recognition of garden biodiversity in local and national Government policy initiatives (Goddard et al. 2010; Loram et al. 2011). There is a consequent need for urban ecologists, planners and policy-makers in particular to understand the current state of the evidence required to develop strategies and approaches to protect and enhance the biodiversity benefits of residential gardens. The present review aims to determine to what extent growing interest in this topic is reflected in studies of garden biodiversity published in the scientific literature, to describe the major themes in this evidence base and identify important evidence gaps.

Literature search

Methods

The literature search and screening process were conducted according to the Preferred Reporting Items for Systematic reviews and Meta-Analyses (PRISMA) guidance (Moher et al. 2009). The Web of Science database was searched (on October 9th, 2022) for articles relating to biodiversity in gardens. Primary search terms relating to gardens were 'garden', 'homegarden', 'yard' and 'backyard'. Each of these was combined with each of the secondary search terms 'biodiversity', 'species richness', 'species diversity' and 'wildlife' in a series of separate searches. These search terms were deemed sufficiently general to pick up most studies relating directly to biodiversity in gardens (the focus of our study), and avoided the less efficient and challenging approach of selecting a sufficiently comprehensive list of taxon-specific terms. The searches were confined to journal articles, reviews and proceedings, but had no date limitations. The results of all the searches were combined and duplicates removed.

The initial list of articles was screened by two of the authors (RJD & DS) by title to sift out those unlikely to be related to private residential gardens. The remaining articles were then screened once more by abstract to remove any not specifically related to biodiversity in private residential gardens. The remainder were classified according to whether they involved the collection and analysis of data at the scale of the individual garden (e.g. plant composition, species diversity), at a broader spatial scale (e.g. by street or neighbourhood) or did not involve data collection (e.g. policy discussion pieces and reviews). Articles were also categorised according to the geographic region they covered (i.e. Africa, Asia, Australasia, Europe, Middle East, North America, South America or Global) and whether they targeted a particular taxonomic group (i.e. plants, invertebrates, birds, mammals, others). We identified studies that provided measures of the diversity of flora and fauna in gardens, that made comparisons with other habitats, and determined whether measures of diversity were related to the characteristics and management of gardens. Finally, we identified any studies that involved experimental manipulations to test predictions related to interventions intended to improve garden biodiversity.

Articles arising from the literature search identified a small number of other sources of information on biodiversity in domestic gardens, and where appropriate these were used to develop themes in the discussion and cited accordingly. Articles identified in this way were not included in the reported bibliometrics.

Results

The initial search of the Web of Science database identified 3774 articles of potential interest. However, this was reduced to 443 articles after the initial screening, and to 408 (published from 1981 to 2022) following the second sift. Screening excluded articles relating to botanical, community and public gardens, allotments, commercial and work yards, studies that were focused on health benefits, ethnobotany, purely socio-economic aspects of gardens, and those with only cursory reference to biodiversity. Publications surviving the sifts included citations of a further seven relevant scientific publications which had not been returned by the original search.

Of the 408 articles identified by the literature search, the majority involved the collection and analysis of data at the scale of the individual garden (77%) rather than at a broader spatial scale (Figs. 1 and 2). However, the latter often included data from individual gardens or evidence of direct relevance to their management. The first published article in 1981 was followed by only seven articles throughout the 1990s, but from 2002 onwards there was a clear upward trend in the number of articles published per year (Fig. 2). The geographic distribution of studies showed a substantial bias towards Europe and North America (32.1% and 23.8% respectively). In contrast, the number of studies from Asia, Africa and South America each accounted for only 9–13% of all publications, the majority of which (68%) focused on home gardens (Fig. 3). The most commonly studied taxonomic groups amongst the 408 articles were plants and invertebrates, with fewer focusing on birds and mammals, whilst other species groups were very poorly represented (e.g. amphibians were the focus of only one study) (Fig. 4). Comparison of the ecological characteristics of gardens to those of

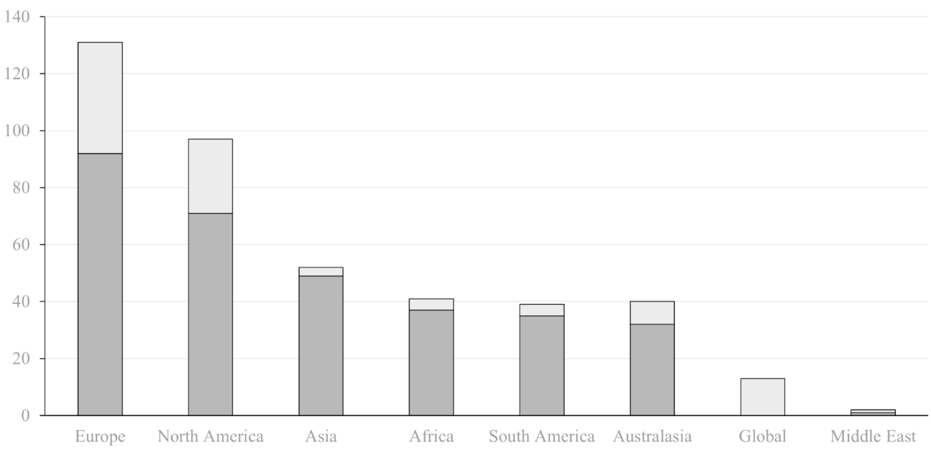

Fig. 1 The number of published articles on garden biodiversity by geographic region, showing the proportion in which field data was collected at the level of the individual garden (dark grey) or at a broader scale (light grey). The totals per region sum to more than the 408 articles identified by the data search as some covered multiple countries

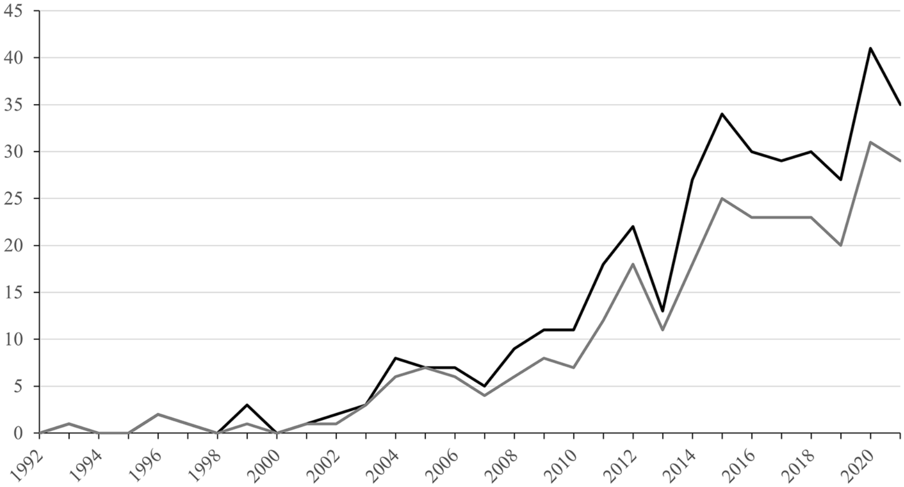

Fig. 2 The total number of published articles relating to gardens and biodiversity by year (black line) and the number of these that involved data collection from individual gardens (grey line). One publication in 1981 and all articles from 2022 (as the search was conducted part-way through that year) have been omitted from the figure

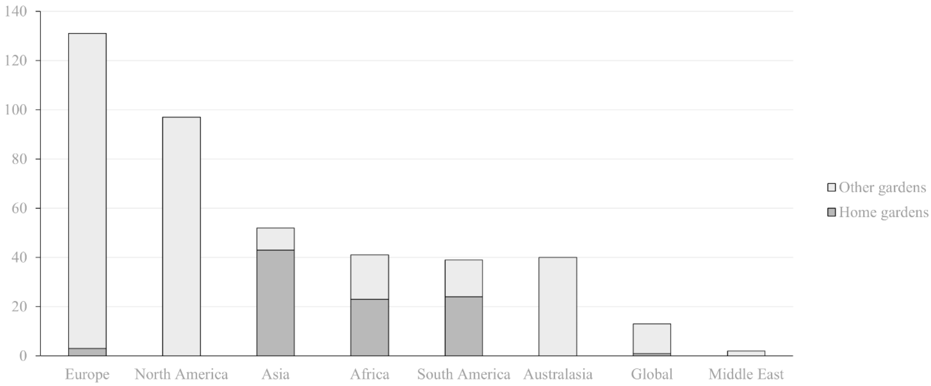

Fig. 3 The number of published articles on garden biodiversity by geographic region, showing the proportion described as 'home gardens' (dark grey). The totals per region sum to more than the 408 articles identified by the data search as some involved multiple countries

other habitats was reported in 86 (21%) publications, most of which (78%) related to plant and invertebrate communities (Fig. 4).

Only 18 articles (4%) described experimental studies conducted in gardens, the majority (72%) of which assessed impacts on invertebrates, particularly pollinators. A further 18 studies involved experiments of direct relevance to garden biodiversity but which were conducted outside of residential gardens (e.g. in experimental facilities).

Of the 296 non-experimental studies involving the collection of ecological data from individual gardens, 73% (n = 217) had calculated species richness or an index of diversity for one or more plant or animal groups. Of these, 54% (n = 116) related at least one measure

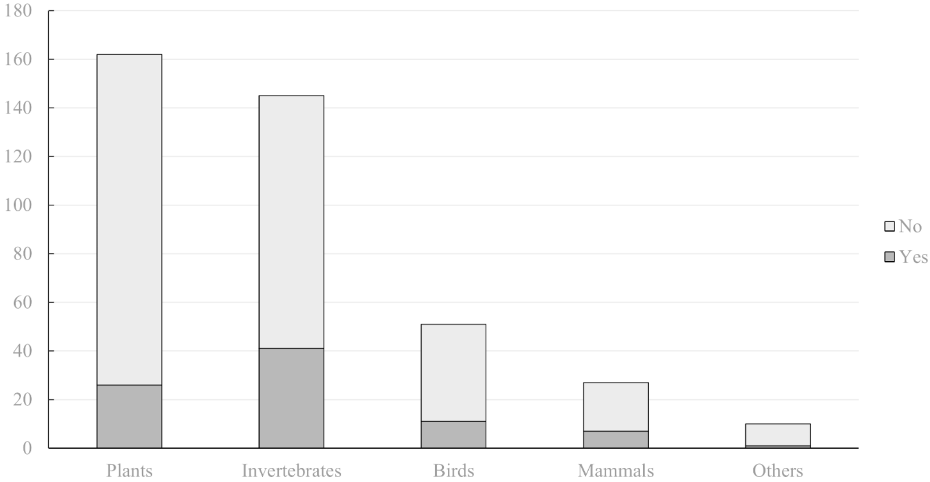

Fig. 4 Number of published studies per taxonomic group showing the proportion that included comparisons between gardens and other habitats (dark grey). The totals per taxonomic group sum to more than the 408 articles identified by the data search as some studies involved more than one group

of biodiversity to the physical characteristics of the garden (total area, vegetation cover, floral composition etc.) but only 13% (n = 29) related them to garden management practices (composting, pesticide use, lawn mowing etc.). However, as garden management practice gives rise to the physical characteristics of gardens (e.g. planting influences floral composition), they are not entirely mutually exclusive terms. The results of these studies could not be directly compared (e.g. in a meta-analysis) as a variety of different measures of biological diversity were employed and parameters to describe the physical characteristics of gardens and management practices varied widely. However, it was possible to discern some broad trends. For example, the most common positive correlate of plant and arthropod species richness was garden size (e.g. Kabir et al. 2009; Loram et al. 2008; Sierra-Guerrero and Amarillo-Suárez 2017; Knapp et al. 2012), whilst the diversity and abundance of invertebrate and vertebrate fauna in gardens was most often positively related to measures of plant and habitat diversity (Otoshi et al. 2015; Quistberg et al. 2016; Simao et al. 2018; Tresch et al. 2019a, b). Invertebrate diversity and abundance were also often positively correlated with the availability of floral resources in the garden (Egerer et al. 2020; Fontaine et al. 2016; Foster et al. 2017; O'Connell et al. 2021; Pardee and Philpott 2014). Garden management practices were expressed in many different ways, and were seldom related to measures of biodiversity, but where they were, lower biodiversity was most often associated with more intense management including frequent lawn mowing, fertiliser and pesticide application (e.g. Lerman et al. 2018; Toledo-Hernández et al. 2016). These themes are explored further in the discussion.

Discussion

General

The literature search clearly identified a substantial upward trend over the last two decades in the number of academic publications related to biodiversity in domestic gardens. This may reflect growing interest in this area and/or simply a more general increase in the number of articles over this period, but the net result has been an expanding body of published evidence relating to biodiversity and domestic gardens. Increasing academic interest in the subject is consistent with growth in wider public interest. Although the literature search was confined to a single bibliographic database and we did not include search terms for specific taxa, publications of potential interest that were cited in the papers returned by the search were often themselves included in the search results (with only seven identified that were not). This provided confidence that the search was sufficiently comprehensive to be able to rely on the trends identified.

The question of whether gardens can make a positive contribution to biodiversity contains an inherent contradiction because the process that usually gives rise to gardens is urbanisation, which is a major cause of global habitat loss and species extinction (Güneralp et al. 2013). Urban environments are also associated with biotic homogenisation (McKinney 2006; Cubino et al. 2020b), a process whereby ecosystems become degraded and simplified, often involving colonisation by non-native species (Klotz and Kühn 2010). Gardens have been implicated in this process through the dispersal of propagules into surrounding habitats (Dutta et al. 2021; McLean et al. 2018). However, an expanding body of evidence emerging in recent years demonstrates that residential gardens have the potential to make a significant contribution to the conservation of biodiversity in the built environment and the provision of ecosystem services (e.g. Cameron et al. 2012; Dewaelheynes et al. 2015) that may extend their influence beyond the urban realm. Furthermore, the relative importance of these benefits is set to grow as the scale of urban landcover increases alongside continuing intensification of agricultural production and degradation of non-urban habitats across the globe. For example, in the UK urban areas are home to a substantial proportion of some bird populations (Gregory and Baillie 1998) whilst Australian cities have been reported to support more nationally threatened animal and plant species than non-urban areas (Ives et al. 2016). Consequently, the question of how to maximise the biodiversity value of gardens has recently gained more prominence in conservation circles (see Larson et al. 2022).

Geographic variation

Results of the literature search suggest that evidence relating to biodiversity in gardens is heavily skewed towards European and North American studies. As our search only included articles with an English language abstract we may have missed some publications written entirely in another language and instances where gardens may have been described using a local colloquialism (e.g. 'patios' in South America), although this is unlikely to completely explain the substantially poorer representation of studies from Asia, Africa and South America. This geographic difference is significant as it is at odds with the scale and rapid pace of urbanisation in these regions (United Nations 2018). A better understanding of the factors that drive variation in garden biodiversity and related ecosystem services in these regions would help inform strategies for sustainable urban development. This should include further consideration of traditional home gardens which often predominate in many parts of south-east Asia, Africa and South America, and which studies indicate may have high plant diversity (Barbhuiya et al. 2016; Huai and Hamilton 2009; Panyadee et al. 2018; Zasada et al. 2020). The diverse plant communities of home gardens have also been associated with an abundance and richness of arthropods (Huerta and van der Wal 2012; Toledo-Hernández et al. 2016), but there is a paucity of information on the potential for them to play a role in wildlife conservation (Webb and Kabir 2009). Home gardens are important sources of food, medicine, cosmetics, spices and other resources for low-income families, and this diversity of uses may drive the relatively high levels of plant diversity often observed (Naigaga et al. 2020). However, several studies have identified a developing trend towards less utilitarian use of home gardens, associated with rising socio-economic status (Bigirimana et al. 2012; Davoren et al. 2016) and urbanisation (Caballero-Serrano et al. 2016; Clarke et al. 2014). Consequent changes in garden composition include an increasing proportion of non-native ornamental plants (Caballero-Serrano et al. 2016) which may boost species richness, although the reported loss of trees and shrubs (Peyre et al. 2006; Poot-Pool et al. 2015) reduces habitat complexity. The relationship between rising household income and reducing utilitarian use of gardens has been reported in studies of domestic gardens elsewhere (e.g. in Jordan (Al-Kofahi et al. 2019) and USA (Martin et al. 2004)) and has been associated with an increase in plant species richness (Avolio et al. 2018, 2020; Van Heezik et al. 2013). Given that home gardens are widespread in many rapidly developing parts of the world, it will be important to understand how changes in their function and structure may influence their contribution to biodiversity conservation and provision of ecosystem services in the face of urbanisation and economic development.

Comparison with other habitats

Of the relatively few studies that compared the biodiversity of gardens to other habitats, several showed similar or higher plant species richness than nearby native habitats including forest, grassland and desert (e.g. Knapp et al. 2012; Pearse et al. 2018). Similar relationships between plant diversity in gardens and semi-natural habitats such as grasslands and scrub were largely driven by the cultivation of non-natives (Baldock et al. 2019; Loram et al. 2008; Thompson et al. 2003). However, scale is important in this regard because although a sample of gardens can collectively yield a remarkably high number of plant species (greater even than that reported for some of the most biodiverse natural habitats), diversity is more comparable with natural habitats at the quadrat scale (Thompson et al. 2003). The species richness of invertebrates in gardens has been reported to be lower than in some natural habitats including native bushland, forests and wetlands (Lowe et al. 2018; Marín et al. 2020; Toft et al. 2019), albeit only marginally in some cases (Fetridge et al. 2008; Lowe et al. 2018). However, observations that gardens enhanced with native plants hosted a greater number of moth species than natural areas such as wetlands and native woodlands (Downer and Ebert 2014) and that garden resources attracted a greater diversity and abundance of mammals than was found in suburban and rural forests (Hansen et al. 2020) illustrate the potential for management practices to buck this trend. Similarly, although a study of gardens in six US cities indicated that they were less diverse than the local native habitats they replaced, those that had been managed for the benefit of wildlife were functionally more similar to native ecosystems (Cubino et al. 2020b).

Drivers of garden biodiversity

To understand how to enhance the biodiversity value of gardens it is necessary to identify the principal drivers of variation. Many studies report a positive correlation between plant diversity and garden area (e.g.Bigirimana et al. 2012; Caballero-Serrano et al. 2016;Kabir et al. 2009; Loram et al. 2008; Sierra-Guerrero and Amarillo-Suárez 2017; Knapp et al. 2012) although for cultivated species this relationship may be weak or absent (Cavender-Bares et al. 2020;Van Heezik et al. 2013; Smith et al. 2006a). This is probably because cultivated plants do not face the same barriers to colonisation as spontaneous native species and can exist as single or few individuals and be replaced as required rather than needing to attain a threshold population size for persistence. Overall, increases in plant diversity may be small relative to large increases in the size of entire gardens (Smith et al. 2006a) and lawns (Thompson et al. 2004). Nevertheless, this positive relationship may relate to the greater invertebrate diversity observed in larger gardens (Burks and Philpott 2017; Fontaine et al. 2016; Quistberg et al. 2016). Complexity of habitat structure has also been cited as a driver of biodiversity in gardens (Bates et al. 2014; Daniels and Kirkpatrick 2006; Smith et al. 2006a; Young et al. 2019) consistent with this heterogeneity supporting a wider variety of ecological niches. One study of invertebrate diversity described taxa-specific variation in the importance of different characteristics of urban gardens, suggesting that the availability of a range of habitats was likely to be important when considering diversity of the entire invertebrate community (Smith et al. 2006b). Similarly, the diversity of mosses and lichens in gardens has been related to the availability of a range of substrates (Oishi 2019; Smith et al. 2010). Unsurprisingly, the extent of vegetation cover in gardens has been positively correlated with the abundance and diversity of arthropods (Lowe et al. 2018; Salisbury 2020; Salisbury et al. 2017) and vertebrates (Van Heezik et al. 2013). Also, the diversity of these plant communities has been reported as an important driver of diversity in several taxa including spiders (Otoshi et al. 2015), bees (Quistberg et al. 2016; Simao et al. 2018; Smith et al. 2006b) and soil invertebrates (Tresch et al. 2019a, b). These positive relationships likely reflect the availability of a greater diversity of food resources (e.g. nectar sources for bees and prey for spiders) and structural complexity providing a wider variety of nesting sites (e.g. for cavity nesting bees) and ecological niches (see also Ebeling et al. 2018). Several studies have identified the importance of floral resources to the abundance and diversity of pollinating insects (Egerer et al. 2020; Fontaine et al. 2016; Foster et al. 2017; Gerner and Sargent 2022; Majewska et al. 2018; O'Connell et al. 2021; Pardee and Philpott 2014) with some evidence that a clustered distribution may permit more efficient exploitation (Plascencia and Philpott 2017).

Habitat heterogeneity in gardens tends to be associated with human perceptions of increased 'wildness' (Rooduijn et al. 2018) as garden habitats come more closely to resemble those of natural ecosystems. Garden management practices that promote formality and 'neatness' on the other hand have been associated with reductions in plant diversity and species richness, largely driven by a scarcity of uncultivated native plants (Cubino et al. 2020a). However, the substantial influence of local factors on arthropod abundance and species richness in urban areas indicates the potential to enhance garden biodiversity through changes in management practices (Otoshi et al. 2015; Philpott et al. 2014; Quistberg et al. 2016) although landscape-scale factors will impose limits on what is possible (Braschler et al. 2020). But there may be some reluctance amongst many gardeners towards adopting a 'wilder' and less formal structure in their gardens. For example, Gaston et al. (2005) could not conduct experimental tests of the biodiversity impacts of refraining from lawn mowing, principally because garden owners were concerned as to how their neighbours would react. Such social pressures may be widespread and are potentially linked to the importance placed on being seen to have apparent 'control' of garden environments (see Larson et al. 2022; Goddard et al. 2013), and perhaps to intrinsic human preferences for environmental characteristics that resemble the savannah (i.e. grasslands with low vegetation and dispersed trees) and aided the survival of early humans (the savannah hypothesis; Balling and Falk 1982). Nevertheless, gardens that tend to be more natural in appearance have in some cases been shown also to have more aesthetic appeal (Lindemann-Matthies and Marty 2013). This apparent paradox perhaps reflects the multiple uses of gardens and the perception that they are an extension of the indoor living area, but it also suggests that strategies to enhance biodiversity in gardens might usefully emphasise the potential simultaneously to boost their attractiveness. Initiatives to improve the biodiversity value of gardens are likely to benefit from further consideration of how this can be reconciled with their aesthetic and recreational benefits (see Larson et al. 2022).

Cultivated non-native species often contribute disproportionately to the diversity of plants in residential gardens, with studies in UK cities indicating they may account for 67–70% of species (Loram et al. 2008; Smith et al. 2006a; Thompson et al. 2003). Although non-native species can be a major threat to native ecosystems (Dueñas et al. 2021), the conservation argument for their broad vilification has been questioned (Davis et al. 2011) and their role in gardens contributes to this debate. There is evidence that many non-native flowering plants may be unattractive to pollinating insects (Garbuzov et al. 2017) with a greater diversity of insect visitors (Rollings and Goulson 2019) and abundance of bees (Pardee and Philpott 2014) in gardens associated with the presence of native species. This may in part relate to flower structure, as in contrast to species that have co-evolved with their pollinators, introduced species and cultivars may have physical characteristics that make them less accessible and hence less attractive to native insects (Comba et al. 1999; Corbet et al. 2001). For example, UK bumblebees were unable to reach the nectar from some non-native plants owing to the depth of the corolla (Corbet et al. 2001). Furthermore, gardens with a proliferation of non-native plants that are unattractive to invertebrates may in turn attract fewer of their vertebrate predators (Narango et al. 2017). Nevertheless, some non-native plants may be important sources of pollen and nectar for many invertebrates (e.g. Garbuzov and Ratnieks 2014; O'Connell et al. 2021), particularly when native species are seasonally unavailable (Koyama et al. 2018; Salisbury et al. 2015), can provide food for their larvae when native host plants have been extirpated (Shapiro 2002) and may be valuable to a range of herbivorous insects (Smith et al. 2006a). Furthermore, experimental studies have confirmed the importance of non-native flowering plants to bees and butterflies despite the availability of native plants (Majewska et al. 2018; Matteson and Langellotto 2011). Hence, the geographic origins of plant species may be less important to enhancing the biodiversity of gardens than their functional attributes, including their contribution to three-dimensional complexity (Davis et al. 2011; Smith et al. 2006a). The potential benefits of some non-native species may have been overlooked in previous sources of advice on 'wildlife gardening' (e.g. Baines 2000; English Nature 2003) but has been recognised in subsequent mainstream advice (e.g. Natural England 2007; RSPB 2022; RHS 2022).

A major theme to emerge from the few studies of the relationship between biodiversity and gardening practices, was a tendency for intensive management to be associated with reduced biodiversity. The intensity of garden management, characterised by regular soil tilling, weeding, watering, fertiliser and pesticide application has been associated with an increase in floral species richness. This appears to have been driven by the cultivation of non-native species, but a decrease in the proportion of native plants (Loram et al. 2011) and of soil fauna biomass, with negative consequences for soil functions such as decomposition, and the retention of water, carbon and nutrients (Tresch et al. 2019a). In a study of home garden biodiversity, lower intensity management (reduced pesticide, herbicide and fertiliser use) was associated with increases in habitat heterogeneity, arthropod abundance and species richness of bees and wasps (Toledo-Hernández et al. 2016). The harmful effects of some pesticides and herbicides on fauna have been well documented and so, unsurprisingly, their use in gardens has been associated with negative impacts on pollinating insects (Muratet and Fontaine 2015; Fontaine et al. 2016), other arthropods (Barratt et al. 2015) and vertebrates (Fardell et al. 2022). Studies have also identified negative impacts on plant species richness in garden lawns, associated with fertiliser application (Cavender-Bares et al. 2020) and watering (Wheeler et al. 2017) as these practices are likely to favour a minority of species which may then predominate. Similarly, frequent lawn mowing may select for species able to reproduce vegetatively, by preventing plants from maturing sufficiently to produce pollen and set seed (Bertoncini et al. 2012). Hence, infrequent lawn mowing has been associated with greater plant diversity (Bertoncini et al. 2012) and subsequent abundance and diversity of bees (Lerman et al. 2018). However, in other studies of lawn composition there were no detectable effects of mowing frequency on plant diversity (Smith et al. 2010; Thompson et al. 2004), and mixed effects in relation to invertebrates which may have reflected taxa-specific responses (Helden et al. 2018). Although reducing the frequency of lawn mowing is often advocated in 'wildlife-friendly' gardening advice, these mixed results suggest the need for more evidence on the biodiversity implications of different mowing regimes. Such studies might usefully address impacts of mowing on plant species form and composition, which could conceivably give rise to fewer resources for some other organisms without necessarily reducing floral species diversity. It may also be beneficial to investigate the potential for regional variation in the responses of vegetation and invertebrates to different mowing regimes. Such information would not only be valuable to underpin advice for residential gardeners but would also be relevant to the management of public lawns in many urban green spaces.

'Wildlife-friendly' gardening

There is no shortage of advice available on wildlife-friendly gardening practices (mostly originating from Europe, North America and Australasia), but evidence for the effectiveness of interventions is relatively scarce and sometimes contradictory. Our literature search identified very few replicated experimental tests of recommended 'wildlife-friendly practices' carried out at scale, despite this evidence gap being identified many years ago (e.g. Gaston et al. 2005). Advice on wildlife-friendly gardening has often tended to favour the planting of native species, and the growth in specialist native plant suppliers may increase their representation in residential gardens (Stewart et al. 2009). Several studies have reported benefits to insect abundance and species richness following the addition of native flowering plants to gardens (Garbuzov and Ratnieks 2014; Griffiths-Lee et al. 2022; Salisbury et al. 2015). Hence, the abundance of monarch butterflies (*Danaus plexippus*) was enhanced through the targeted provision of their native milkweed (*Asclepias* spp.) larval host plants (Cutting and Tallamy 2015). However, such benefits have not been universally reported following the small-scale addition of native flowering plants to gardens (Matteson and Langellotto 2011). Variation amongst studies may in part relate to differences in the nectar production and attractiveness of plants at varying stages of development and under different growing conditions (Rollings and Goulson 2019). These studies illustrate the challenges in generating generic recommendations. It is no surprise therefore that advice on 'pollinator-friendly plants' appears fragmentary, as illustrated by a review of available lists which showed that although most species included were beneficial, there were low levels of overlap and some included species of little benefit and either omitted those known to be attractive to flower-visiting insects or did not rank them highly (Garbuzov and Ratnieks 2014; Rollings and Goulson 2019). Future research could usefully address potentially confounding factors in studies through experimental design, although controlling for locally varying factors (such as the characteristics of adjacent habitats and connectivity with them) is likely to remain challenging for any garden-based studies. Also worthy of further research are the broader implications for resident native plant species of sowing wild flowers (e.g. hybridisation and competition), which remain unclear (Johnson et al. 2017).

Mixed results were obtained from a study by Gaston et al. (2005) to test the efficacy of a variety of 'wildlife gardening' interventions with some approaches being demonstrably successful (e.g. provision of small ponds) whilst others exhibited low rates of success in the short to medium term (e.g. nettle patches and artificial bumblebee nests). This is likely a reflection of the effects of the scale of interventions (some may have low success at the scale achievable/desirable within typical gardens), of the context within which they are conducted (e.g. being less successful when conducted in isolated rather than in multiple gardens, or in gardens where colonisation or use by organisms from elsewhere is unlikely), and of time (as some interventions may take longer to be successful). Nonetheless, although little evidence exists for their effectiveness, a wide range of products now exists for the 'wildlife-friendly' gardener. The few studies to have investigated commercially available invertebrate nesting aids (also called 'bug hotels') report relatively low levels of occupancy by some taxa, including native bees, and concerns regarding design (Harris et al. 2021; von Königslöw et al. 2019) and their potential to attract parasites and predators (MacIvor and Packer 2015). In one study, nesting aids designed and tailor-built using information on the nesting requirements of Hymenoptera were shown to out-perform a commercially available product (von Königslöw et al. 2019), thus demonstrating the clear need for product design to be informed by empirical evidence. In contrast, the appearance of many 'wildlife-friendly' products (e.g. bird nest boxes and 'bug hotels') suggests that they may be designed to appeal to the perceived aesthetics of potential purchasers over practical effectiveness. There is a clear need for more longer-term, replicated studies carried out at scale and using standardised approaches to investigate the effectiveness of a range of interventions on garden biodiversity.

The conservation benefits of gardens

Most studies of biodiversity in gardens focused on plants and invertebrates, followed by birds and mammals, but with other taxa being less well represented. The potential importance of gardens for wildlife conservation was identified by several studies. In the UK for example, domestic gardens may be important habitats for mammals such as the nationally declining Western European hedgehog (*Erinaceus europaeus*) (Baker and Harris 2007; Williams et al. 2015). Furthermore, feeding birds in gardens has been reported to have had significant positive impacts on several species (Plummer et al. 2019) and wider benefits for nature conservation (Reynolds et al. 2017). Nevertheless, the potential for supplementary provisioning to restructure bird communities (Galbraith et al. 2015) and to have wider ecological impacts including encouraging dependency on supplementary food, and enhancing competitive interactions, local predation and disease transmission, have also been highlighted (see Shutt and Lees 2021). In some circumstances gardens may be critically important for species conservation, as illustrated by the endangered Western ring-tail possum (*Pseudocheirus occidentalis*) which almost exclusively uses residential gardens of the Australian suburbs (van Helden et al. 2020). Similarly, South African suburban gardens have been colonised by the endangered Knysna warbler (*Bradypterus sylvaticus*) as their vegetation structure mimics that of the disappearing woodland glades of its natural habitat (Pryke et al. 2011). Also, the supplementary feeding of reintroduced red kites (*Milvus milvus*) in UK gardens was identified as a major factor in the recovery of this previously locally extinct species (Orros and Fellowes 2015). Studies of amphibians in gardens were very poorly represented in the literature search results. This seems paradoxical given the substantial global decline in this group (Stuart et al. 2004) and the popularity of garden ponds. The potential value of ponds to amphibians and other taxa (Hill et al. 2015; Hill and Wood 2014) suggests a clear need for evidence to inform effective garden pond management advice (Hill et al. 2021). Gardens may also provide valuable habitat for native plants that are declining elsewhere (Doody et al. 2010; Larios et al. 2013; Rooduijn et al. 2018; Segar et al. 2022), including nationally and internationally endangered species, as illustrated by the presence in home gardens of Brazilwood (*Caesalpinia echinata*) the national tree of Brazil (Akinnifesi et al. 2010) and the Eastern Cape blue cycad (*Encephalartus horridus*) of South Africa (Lubbe et al. 2011). In some areas the plant communities of home gardens represent relics of the former surrounding natural habitats (Blanckaert et al. 2004) or important repositories of genetic resources in the form of scarce traditional crop varieties (Barbhuiya et al. 2016; Galluzzi et al. 2010; Salako et al. 2014).

Residential gardens may also provide indirect benefits to conservation through their capacity to elicit positive human behaviours. Garden-based citizen science projects for example, have not only collected valuable information on the distribution and abundance of species (e.g. birds (Cannon et al. 2005); butterflies (Fontaine et al. 2016); moths (Bates et al. 2014); ants (Lucky et al. 2014); mammals (Toms and Newson 2006)), but have also been linked to changes in participant behaviour that benefit biodiversity. Hence, involvement in garden butterfly surveys was associated with participants providing more nectar resources and reducing pesticide use (Deguines et al. 2020). Public engagement in such schemes may also have wider benefits for biodiversity by increasing knowledge and nurturing pro-conservation attitudes (Cosquer et al. 2012; Evans et al. 2005), which may be particularly important as opportunities to connect with nature become more difficult to find for an increasingly urbanised population.

Challenges and opportunities

No garden is an island, as exemplified by the importance of the surrounding local environment as a driver of biodiversity in individual gardens (e.g. Smith et al. 2006b), particularly for more mobile species (Ellis and Wilkinson 2020) and the dispersal of plant propagules (Stewart et al. 2009). Although individual gardens are often small and highly fragmented, they collectively account for a significant proportion of urban green infrastructure (Loram et al. 2007). Furthermore, gardens may serve to connect various urban green spaces to one another (Rudd et al. 2002) and join up tree canopy cover (Ossola et al. 2019), so making a greater contribution to urban biodiversity than the sum of their parts. Testimony to this is the demonstrated value of gardens as dispersal corridors (e.g. bats (Mimet et al. 2020); shrews (Vergnes et al. 2013); arthropods (Pla-Narbona et al. 2021; Vergnes et al. 2012)). Correspondingly, several studies have described how physical barriers in urban environments may reduce connectivity thus hindering the dispersal of invertebrates (Parris 2006; Peralta et al. 2011), mammals (Hof and Bright 2009) and amphibians (Hamer and Parris 2011). Studies of urban bird populations suggest that residential gardens may collectively function as contiguous habitat when connected to one another and to other habitat patches (Andersson and Bodin 2009; Cox et al. 2016). Melles et al. (2003) observed patterns of habitat use consistent with birds supplementing resources in small urban habitat patches in residential areas (e.g. gardens) with those from larger patches of nearby habitat (e.g. parks). The concept of ecological land-use complementation is relevant in this context as it describes how clusters of habitat patches may provide different but complementary resources (e.g. food, nesting or roosting sites) thereby forming ecologically functional units (Colding 2007). It follows that to maximise the biodiversity benefits of residential gardens they need to be managed in inter-connected clusters with links to other habitats. The design of new urban developments could therefore usefully include features that promote rather than hinder biological connectivity amongst gardens and other green spaces (e.g. hedgerows (Dixon 2022)). Network approaches might be a useful tool for identifying connections amongst gardens and other urban green infrastructure that maximise biodiversity benefits (e.g. Rudd et al. 2002). The planning of residential developments might also usefully incorporate variation in the composition of new gardens to enhance habitat heterogeneity (Gaston et al. 2007), and favour retention of existing trees to offset biodiversity losses (Reis et al. 2012) and enhance connectivity of canopy cover (Ossola et al. 2019). To date much research on the biodiversity value of gardens has involved studies on individual gardens, but in order to better understand and manage them as inter-connected habitats it may be useful to adopt a landscape ecology approach (Goddard et al. 2010).

Several studies reported trends in garden land use that present challenges for urban biodiversity. For example, the continuing loss of front gardens to car parking space and of rear gardens to development (Smith et al. 2011; Laćan et al. 2020) not only reduces biodiversity but also removes associated ecosystem services such as carbon sequestration and storage, and flood control (Alexander 2006; Warhurst et al. 2014). Concern over the replacement of gardens with hard surfaces has led to legislative changes to UK planning policy (Communities and Local Government 2008) although their effectiveness and the ability of planning authorities to enforce them is debatable. Hence, from 2005 to 2015 the number of front gardens in the UK that were replaced with hard surfaces tripled, to a point where they accounted for an estimated 54% of the total surface area of all front gardens (RHS 2016). Also, changes in UK Government policy intended to reduce urban sprawl resulted in the loss of urban green space (including gardens) to development (Dallimer et al. 2011), with 39% of Local Planning Authorities in England citing residential development in gardens as a significant issue (Sayce et al. 2012). Trends towards increasing housing density reduce average individual garden plot size, and although the cumulative area of garden habitat may remain the same, connectivity could be impaired by the presence of more barriers to species dispersal. Even urban-adapted species may be at risk from trends in residential development, as indicated by the negative impact of reduced vegetation cover in new housing developments on their attractiveness to house sparrows (Moudrá et al. 2018). Increased tree, hedgerow and shrub planting in new developments is an example of where the planning system may be the most appropriate vehicle for urban biodiversity enhancement. However, ease of implementation and cost remain dominant considerations for many developers in the gardens that they create, and so more consideration could be given to the best advice to provide them with.

The design and management of publicly owned green infrastructure can be steered towards benefiting biodiversity through the implementation of public policy. Similarly, planning systems may be able to exert positive influence on the design of garden spaces in new residential developments in the interests of biodiversity, promoting features that are consistent with inter-connectivity for example. However, the characteristics and routine management of private gardens are the consequence of many individual preferences and decisions (Kendal et al. 2012) influenced by multiple socio-economic factors and motivations (Cavender-Bares et al. 2020; Lowenstein and Minor 2016; Philpott et al. 2020) and so may be less easily influenced by public policy. One approach that has improved knowledge and stimulated the adoption of beneficial gardening practices is engagement with householders to encourage self-assessment of the biodiversity value of their gardens along with provision of specialist advice and positive feedback (Van Heezik et al. 2012). Collaborations between local public authorities and private households (Mumaw and Bekessy 2017) or local neighbourhood associations (Lerman et al. 2012; Larson et al. 2022) may be useful vehicles for such schemes. Another approach is to provide advice and incentivise uptake by offering an endorsement of the habitat created, as demonstrated by the success of the US National Wildlife Federation's Certified Wildlife Habitat Scheme (Widows and Drake 2014). Observations of social contagion in gardening behaviour (Hunter and Brown 2012) suggest that such local initiatives may gain momentum once a certain proportion of occupants in an area are engaged (Van Heezik et al. 2012) and this may be important in creating clusters of biodiverse gardens. The further development of novel strategies to enhance the biodiversity of residential gardens is likely to benefit from research into how individual choices in garden management are shaped by social and environmental factors (Marco et al. 2010) and how managing gardens in the interests of biodiversity can be reconciled with their other functions (Elliot Noe et al. 2021).

Conclusions

Gardens are undoubtedly an important source of biodiversity and ecosystem services, particularly in urban areas, and facilitate people's connection with the natural world that may promote environmental awareness. Maximising the biodiversity benefits of domestic gardens is becoming increasingly important as urban areas expand and biodiversity in the wider environment continues to decline. Biodiversity in these designer ecosystems is driven by a range of interacting factors related to the local environment and human management. Although academic interest in garden biodiversity has grown substantially in the last two decades, the associated evidence base is heavily biased towards studies of plants and invertebrates in Europe and North America. In contrast there is limited evidence relating to other parts of the world where the pace of urbanisation may be particularly rapid. Another evidence gap relates to the role of gardens in the conservation of certain taxa such as reptiles and amphibians for which there are very few studies. There is also a paucity of empirical evidence from replicated experimental studies to indicate what management interventions work best and under what circumstances. Enhancing the biodiversity value of gardens will also require initiatives to maintain connectivity amongst gardens and with other green infrastructure, and to promote low intensity management and habitat heterogeneity at the level of the individual garden, whilst managing emerging threats to gardens such as their replacement by development, conversion to hard surfaces and declining plot sizes. Meeting these challenges will require greater recognition of the biodiversity value of gardens amongst policymakers and planners, public participation in co-ordinated local garden biodiversity initiatives and arguably more responsive regulation.

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Data Availability The dataset generated and analysed during the current study is available at https://data.mendeley.com/datasets/97fvwbwwbh/3.

Declarations

Competing interests The authors declare no competing interests.

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