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REVIEW

The future of urban agriculture and biodiversity-ecosystem services: Challenges and next steps

Brenda B. Lin[a,*], Stacy M. Philpott[b], Shalene Jha[c]

[a]*CSIRO Climate Adaptation Flagship PMB 1, 107-121 Station Street, Aspendale 3195, VIC, Australia*
[b]*Environmental Studies Department, 1156 High Street, Santa Cruz, CA 95064, USA*
[c]*Integrative Biology, University of Texas at Austin, 401 Biological Laboratories, 1 University Station C0930, Austin, TX 78712, USA*

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Abstract

Urban landscapes are spatially constrained, and vegetative land uses that provide beneficial ecosystem services are difficult to maintain. Urban agricultural (UA) systems appear in many forms – from community farms and rooftop gardens to edible landscaping and urban orchards – and can be productive features of cities and provide important environmental services. As highly managed plant communities, UA can exhibit high levels of biodiversity, often exceeding that of other green space areas within the city. Additionally, it is likely that variation in vegetation cover, diversity, and structure influence not only the biodiversity in UA, but also the quantity and quality of ecosystem services supported by such systems. The biodiversity and ecosystem services (B&ES) of UA can have potentially large societal and environmental benefits for cities, such as enhanced food security, air quality, and water regulation. Yet few studies have synthesized knowledge regarding UA vegetation management impacts on the quantity, quality, and stability of B&ES provided. This article presents the first survey of the existing research on the characteristics of UA management and their potential to support ecosystem service delivery. Specifically, we examine: (1) biodiversity patterns in UA, (2) ecosystem services provided by UA, and (3) the challenges of promoting UA systems that support B&ES. Overall, our review reveals that varied vegetative structure, increased native plant diversity, and reduction of urban impervious surface are key features of UA systems that contribute significantly to urban biodiversity and provide important ecosystem services such as pollination, pest control, and climate resilience. We conclude with a discussion of critical gaps in current research and strategies to better understand and support UA and ecosystem services.

Zusammenfassung

Urbane Landschaften sind räumlich eingegrenzt, und Landnutzungen durch Pflanzungen, die nützliche Ökosystemdienstleistungen erbringen, sind schwer zu unterhalten. Systeme der urbanen Landwirtschaft (UL) erscheinen in vielerlei Formen -von Gemeinschaftshöfen und Dachgärten bis hin zu ‘edible landscaping’ und urbanen Obstplantagen- und können produktive Elemente in Städten sein und wichtige Umweltdienstleistungen erbringen. Als intensiv bewirtschaftete Pflanzengemeinschaften kann UL reiche Biodiversität aufweisen, die oft die von anderen urbanen Grünzonen übertrifft. Zudem ist es wahrscheinlich, dass die Variation der Vegetation, Diversität und Struktur nicht nur die Biodiversität in der UL beeinflusst, sondern auch Quantität und Qualität der Ökosystemdienstleistungen, die von solchen Systemen erbracht werden. Die Biodiversitäts- und Ökosystemdienstleistungen der UL können potentiell großen Nutzen für Gesellschaft und Umwelt der

*Corresponding author. Tel.: +61 3 9239 4476; fax: +61 3 9239 4444.
E-mail addresses: Brenda.Lin@csiro.au, bbclin@gmail.com (B.B. Lin), sphilpot@ucsc.edu (S.M. Philpott), sjha@austin.utexas.edu (S. Jha).

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Städte haben, z.B. bessere Nahrungsv ersorgung, Luftqualität und Wasserregulation. Nur wenige Studien haben indessen den Wissensstand zum Einfluss des Managements der Vegetation in der UL auf die Quantität, Qualität und Stabilität der erbrachten Biodiversitäts- und Ökosystemdienstleistungen zusammengefasst. Dieser Artikel bietet den ersten Überblick über die Forschung zu den Merkmalen der UL und ihrem Potential Ökosystemleistungen zu unterstützen. Im Einzelnen untersuchen wir die Biodiversitätsmuster in der UL, die erbrachten Ökosystemdienstleistungen und die Schwierigkeiten bei der Förderung von UL-Systemen, die Biodiversitäts- und Ökosystemdienstleistungen unterstützen. Insgesamt zeigt unser Review, dass eine vielfältige Vegetationsstruktur, erhöhte Pflanzendiversität und die Reduktion von undurchlässigen urbanen Bodenoberflächen die Schlüsselfaktoren sind, die UL signifikant zu urbaner Biodiversität beitragen lassen und für wichtige Ökosystemdienstleistungen wie Bestäubung, Schädlingskontrolle und ein ausgeglicheneres Stadtklima sorgen. Abschließend diskutieren wir die kritischen Lücken der gegenwärtigen Forschung und Strategien, um die UL und ihre Ökosystemdienstleistungen besser zu verstehen und zu unterstützen.

Keywords: Food security; Urban planning; Vegetation complexity; Agricultural management; Gardens; Green space

Introduction

Urbanization is a major driver of land cover change worldwide (Grimm et al., 2008) and affects the biophysical and socioeconomic landscape. It is estimated that by 2030 >60% of the global population will live in urban areas (United Nations, 2005). Furthermore, in many parts of the world, human development is expanding rapidly at the edge of urban areas (Brown, Johnson, Loveland, & Theobald, 2005) and the quality of rural habitat is declining owing to agricultural intensification (Benton, Vickery, & Wilson, 2003). Thus, green spaces found within urban landscapes are quickly becoming important refuges for native biodiversity (Goddard, Dougill, & Benton, 2010).

Urban planners are increasingly interested in maintaining agriculture within and around cities due to food security concerns. Many cities contain ‘food deserts’, where access to fresh produce is limited due to reduced proximity to markets, financial constraints, or inadequate transportation (Thomas, 2010; ver Ploeg et al., 2009). In response to food insecurity, urban agriculture (UA) has expanded rapidly. For example, in the US, UA has expanded by >30% in the past 30 years, especially in under-served communities (Alig, Kline, & Lichtenstein, 2004). This is because urban agriculture can be very productive, providing an estimated 15–20% of the global food supply (Hodgson, Campbell, & Bailkey, 2011; Smit, Nasr, & Ratta, 1996), and cities can provide good infrastructure, access to labor, and low transport costs for local food distribution (Hodgson et al., 2011). Additionally, interest in UA has escalated recently due to the desire to transform vacant land in post-industrial cities and to address nutrition and childhood obesity issues in disadvantaged urban neighborhoods (Yadav, Duckworth, & Grewal, 2012).

Though public and scientific interest in UA has grown dramatically in the past two decades, there are still significant challenges for integrating UA in an increasingly spatially constrained urban landscape. Much of the debate is centered on land-use trade-offs of UA versus other types of urban development. Although there are a number of socioeconomic considerations that affect the development and proliferation of UA in cities, this review will focus on the ecological aspects of the UA system and how they can be designed to maximize the environmental and health benefits in order to increase acceptance of this particular land use in the urban sphere.

One way to encourage the integration of UA is to better understand how planned and associated biodiversity within these systems contribute to urban ecosystem services. However, there are three major gaps in the literature regarding UA status and impacts that limit our ability to increase the range of benefits and ecosystem services that could come from UA systems. First, biodiversity patterns in urban agroecosystems have only recently been documented (Beniston & Lal, 2012) and require synthesis. Second, ecological communities within UA may translate to the delivery of valuable ecosystem services (e.g., pollination, pest-control, water regulation) (Daily, 1997); however, the availability of these services within UA has not been well-established. Finally, little is known about the role of UA in mediating resilience to major threats, specifically climate variability (Eriksen-Hamel & Danso, 2010). Considering the potential benefits of UA for improved ecosystem functioning, in this review we discuss (1) the ability of UA to support local and landscape level biodiversity, (2) the role of UA in providing ecosystem services, and (3) the agenda for future UA research.

What is urban agriculture?

Urban agriculture (UA) is defined as the production of crop and livestock goods within cities and towns (Zezza & Tasciotti, 2010), generally integrated into the local urban economic and ecological system (Mougeot, 2010). UA can include the peri-urban agricultural areas around cities and towns, which may provide products to the local population (Mougeot, 2010).

Fig. 1. Photographs of different types of UA. (a) Community garden in Toledo, Ohio, (b) Allotment garden in Salinas, California, (c) Private garden in Toledo, Ohio, (d) Easement garden in Melbourne, Australia, (e) rooftop garden in New York, New York, (f) urban orchard in San Jose, California. Photos courtesy of P. Bichier (a, b, f), P. Ross (c), G. Lokic (d), and K. McGuire (e).

Urban agriculture activities are diverse and can include the cultivation of vegetables, medicinal plants, spices, mushrooms, fruit trees, ornamental plants, and other productive plants, as well as the keeping of livestock for eggs, milk, meat, wool, and other products (Lovell, 2010). This definition points to the fact that UA is not solely for food production, but for a wide range of needs of the local community, including medicinal and ornamental plants. The different types of UA allow for a diverse set of vegetation structures to contribute to the edible landscape in a range of community types (McLain, Poe, Hurley, Lecompte-Mastenbrook, & Emery, 2012), and this wide range of products means that UA systems are highly heterogeneous in size, form, and function.

A description of different types of UA can be found in Table 1, with associated images in Fig. 1. *Community or allotment gardens* often represent small-scale, highly-patchy, and qualitatively rich semi-natural ecosystems and are usually located in urban or semi-urban areas for food production (Colding, Lundberg, & Folke, 2006). *Private gardens* are primarily located in suburban areas and may be the most prevalent form of urban agriculture in cities (Loram, Tratalos, Warren, & Gaston, 2007). Privately owned gardens cover an estimated 22–27% of the total urban area in the UK (Loram et al., 2007), 36% of urban area in New Zealand (Mathieu, Freeman, & Aryal, 2007), and 19.5% of the urban area in Dayton, Ohio, USA (Sanders & Stevens, 1984). *Easement gardens* are located within private/community properties, but are often regulated by the local government (Hunter & Brown, 2012). Urban easements are established with the purpose of improving water quality and erosion control (Forman & Alexander, 1998), but they can include a wide array of biodiversity, depending on management type (Hunter & Hunter,

Table 1. A summary of the different types of urban agriculture and the potential ways that biodiversity and ecosystem services (B&ES) can be enhanced in each system.

Urban agriculture type	Description	Strategies for B&ES enhancement
Allotment or community gardens	Areas in cities reserved for non-commercial horticulture, containing small garden plots with individual or family management rights to land. In allotment gardens, land is sub-divided and parcels are cultivated individually. In community gardens, an entire area is tended by a collective group.	• Increased sunlight and floral area increases bee and butterfly species richness • Production of local and cultural vegetables as well as medicinal plants can increase the vegetative complexity and diversity of UA systems • Floral diversity and prolonged growing seasons support urban pollinators, seed dispersal, and pest regulation to the broader landscape • Support for below-ground invertebrates and microbes control soil-dwelling stages of insect pests • Facilitate drainage and reduce urban heat island effect
Private gardens	A multispecies production system on the area of land around the house to meet different physical, social, and economic needs and functions; is traditionally an important land use activity for individual households. They may comprise a few square meters of multi-layered diverse vegetation to the other extreme where there may be a large area of single dimension paving with no vegetation at all.	• Stratified vegetation in home gardens can support large amounts of planned and associated biodiversity • Native plantings can increase bird and butterfly diversity • Parasitoid diversity increases with floral diversity • Garden size and 3D structure increase mammalian species groups abundance and diversity • Genetic diversity improves the connectivity of threatened and rare species
Easement gardens	Clustered in small patches across many different types of neighborhoods in unused vegetative patches next to road ways. These areas can be transformed and improved from simple grass strips areas into vegetatively diverse community orchards and gardens that better support biodiversity.	• Increased vegetation cover can improve water quality and reduce soil erosion • Improves matrix for species movement between land use types • Increased vegetation can provide habitat and help support biodiversity • Urban street trees can contribute to the cooling of adjacent buildings • Noise and environmental buffers along rail and highway corridors
Roof-top gardens	Established on the roof of a building. Farming is usually done using green roof, hydroponics, aeroponics or air-dynaponics systems, or container gardens techniques.	• Provide wildlife habitat for pollination and pest control • Reduced impervious service increase air cooling, flood mitigation, and wildlife habitat
Community orchards	Landscape of tall trees changing with the seasons, fruit of many kinds, good soil, and an array of wild life. Owned and run by the community, some by local authorities with local people.	• Provide wildlife habitat • Provide dense tree structure for carbon sequestration in trees and soil • Increased vegetative cover helpful for erosion control and storm attenuation

2008). *Roof-top gardens* are any garden established on the roof of a building and can be both decorative and used for agriculture. *Urban orchards* are tree-based food production systems that can be owned and run privately or by the community. Increasingly, schools and hospitals are establishing fruit trees that provide crops, erosion control, shade, and wild life habitat, while producing food for the local community (Drescher, Holmer, & Iaquinta, 2006). Many UA systems may fit into more than one category. For example, both private gardens and community gardens may exist as rooftop gardens, and community orchards may exist within community gardens.

Urban agricultural systems and biodiversity provision

Urban landscapes are typically highly simplified, intensively developed ecosystems with low levels of native biodiversity (Lin & Fuller, 2013). However, urban green spaces such as UA can bring diverse green infrastructure back into the urban system, providing vegetative structure and biodiversity for ecosystem function and services across fragmented habitats and spatial scales (Lin & Fuller, 2013). UA may be especially important for biodiversity conservation in cities because vegetative structural complexity in simplified

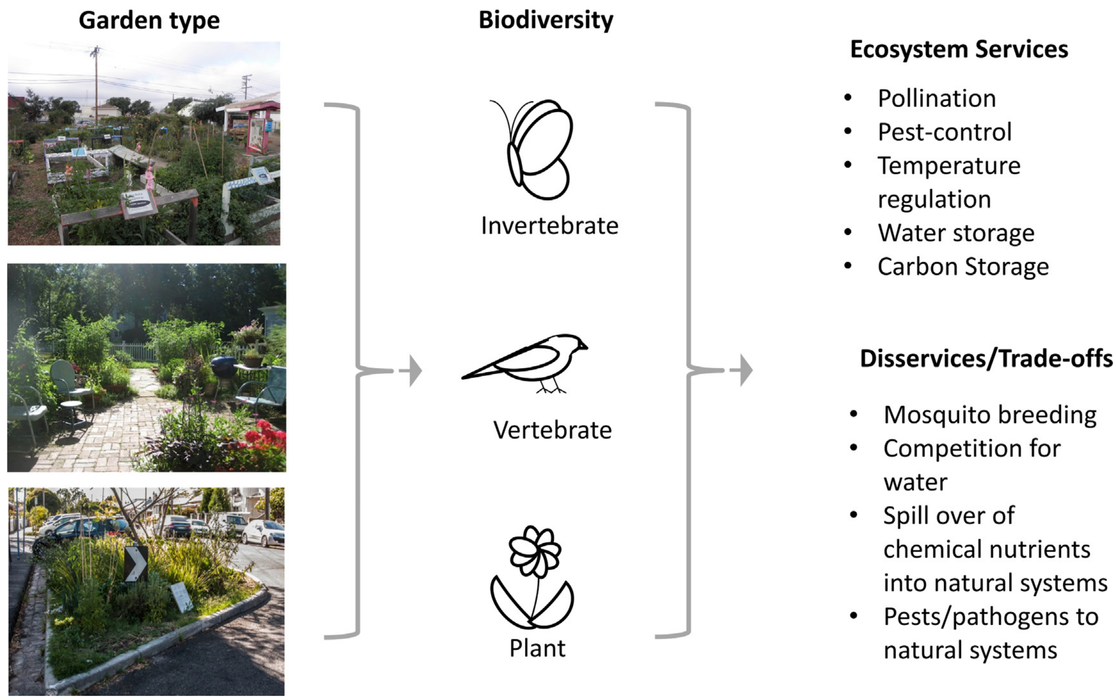

Fig. 2. Conceptual diagram of how different types and management of urban agriculture may lead to different configurations of biodiversity and both ecosystem service benefits as well as trade-offs.

landscapes (e.g. cities) contributes disproportionately more to conservation than in more natural landscapes (Tscharntke et al., 2012). Just as in agricultural systems where more complex agri-environment management can have a larger effect on biodiversity when implemented in simple agricultural landscapes than more complex landscapes (Tscharntke et al., 2012), UA provides many opportunities for re-vegetating the landscape at the local scale within a vegetatively depauperate urbanized landscape.

Urban agriculture has the potential to support biodiversity not only within UA sites, but also nearby due to a landscape-mediated 'spill over' of energy, resources, and organisms across habitats. Such spill over may be an important process for the persistence of wildlife populations in human-dominated landscapes because it allows for resource acquisition and re-colonization events (Blitzer et al., 2012). At the same time, chemical, water, and animal movement is bi-directional, and intensified management implemented in backyards, such as pesticide application, extensive pruning, frequent mowing and other disturbances, can limit the capacity of gardens to maintain rare or sensitive insect species (Matteson & Langellotto, 2011). Thus, it is important to understand that not all biodiversity is necessarily "good" biodiversity, and there may be a number of disservices that come from UA as well (discussed further in the Ecosystem services provision to agriculture and the broader landscapes section). Here, we examine the ability of UA systems to support vegetative, insect, and vertebrate biodiversity (Table 1, Fig. 2).

Vegetative diversity

The wide variety of UA types in practice allow for considerable variation in vegetative complexity and diversity. Domestic gardens vary widely in features that may promote plant biodiversity, such as ponds, moss, groundcover, and varied vascular vegetative structures (Smith, Gaston, Warren, & Thompson, 2005). For example, tropical home gardens have stratified vegetation similar to those seen in multi-stratified agroforestry systems (sensu Moguel & Toledo, 1999) and can thus provide a large amount of planned and associated biodiversity (WinklerPrins, 2002). The diversity of vegetation types within home gardens has been documented in Santarem, Brazil, where 98 plant species were identified in 21 urban gardens and included a large diversity of fruit trees and shrubs (comprising 34% of garden cover), ornamental plants (10%), vegetable/herb plants (13%), and medicinal plants (45%) (WinklerPrins, 2002). In Leon, Nicaragua, 293 plant species belonging to 88 families were recorded across 96 surveyed home gardens, ranging in habit and taxonomic origin (González-García & Sal, 2008). In Hobart, Australia, 12 distinctly different garden types with different species, habits, and canopy heights were documented in front and backyard gardens (Daniels & Kirkpatrick, 2006a), and a similar survey conducted in Toronto found 25 woody plant species and 17 different herbaceous plant species per backyard garden (Sperling & Lortie, 2010). In an example from five UK cities, more than 1000 species were recorded in 267 gardens, exceeding that recorded in all other local urban and semi-natural habitats (Loram, Warren, & Gaston, 2008).

Allotment and community gardens also provide substantial levels of vegetative biodiversity. In Stockholm, allotment gardens are older than many backyard gardens, often representing lush, well-managed flower-filled areas ranging in size (3450–70,000 m^2). Such areas are often extremely rich in plant diversity, with more than 440 different plant species recorded in a single 400 m^2 allotment garden (Colding et al., 2006). In Toronto, surveys showed that besides the typical local vegetables (cabbage, tomatoes, peppers, and eggplant), farmers grew an additional 16 vegetable crops to supply the local community with foods unavailable in local grocery stores. These crops included Asian vegetables, such as bok choy, long bean, hairy gourd, and edible chrysanthemums and substantially increased the vegetative diversity of the urban garden system (Baker, 2004).

Insect diversity

Plant diversity is a principle predictor of insect diversity at small spatial scales (Southwood, Brown, & Reader, 1979), and plant diversity and small-scale structural complexity is important for tree-dwelling arthropods (Halaj, Ross, & Moldenke, 2000), ground-dwelling arthropods (Byrne, Bruns, & Kim, 2008), web spiders (Greenstone, 1984), grasshoppers (Davidowitz & Rosenzweig, 1998), bees (Jha & Vandermeer, 2010), and ground-dwelling beetles (Romero-Alcaraz & Ávila, 2000). However, arthropod species richness has been shown to decrease with increasing impervious surface and intensive management in urban green areas, and intensive UA would presumably have a negative impact of species richness (Sattler, Duelli, Obrist, Arlettaz, & Moretti, 2010).

In a study of urban backyard gardens in Toronto, invertebrate abundance and diversity was enhanced as the number of woody plant structures and plant species diversity increased, and backyard gardens had higher abundances of winged flying invertebrates when compared with urban grasslands and forests (Sperling & Lortie, 2010). Likewise, within domestic gardens in the UK, invertebrate species richness was positively affected by vegetation complexity, especially the abundance of trees (Smith, Warren, Thompson, & Gaston, 2006). In Pennsylvania, butterfly diversity increased with native plantings within suburban gardens (Burghardt, Tallamy, & Gregory Shriver, 2009), and parasitoid diversity increased with floral diversity within urban sites (Bennett & Gratton, 2012).

Because allotment gardens often exhibit a rich abundance of flowering plants and thus a prolonged season for nectar supply, allotment gardens can support urban pollinators for long periods of time (Colding et al., 2006). In a survey of 16 allotment gardens in Stockholm, the number of bee species observed per allotment garden ranged between 5 and 11, including a large number of bumble bees, which were observed on a total of 168 plant species, especially those in the Lamiaceae, Asteraceae, Fabaceae, Boraginaceae and Malvaceae (Ahrne, Bengtsson, & Elmqvist, 2009). In a survey of gardens in Vancouver, a mean richness of 23 bee species were found across the different garden types sampled (Tommasi, Miro, Higo, & Winston, 2004). Similarly, community gardens in NYC were found to provide a range of ornamental plants and food crops that supported 54 bee species, including species that nest in cavities, hives, pith, and wood (Matteson, Ascher, & Langellotto, 2008). In another study in NYC community gardens, the authors found that butterflies and bees responded to sunlight and floral area, but bee species richness also responded positively to garden canopy cover and the presence of wild/unmanaged areas in the garden (Matteson & Langellotto, 2010). In Ohio, bee abundance in private, backyard gardens increased with native plantings, increases in floral abundance, and taller herbaceous vegetation (Pardee & Philpott, 2014). Overall, these studies support the idea that UA management with high vegetation diversity can have positive effects on invertebrate biodiversity in urban systems.

Vertebrate diversity

Wildlife friendly features implemented in gardens can increase vertebrate diversity (Goddard, Dougill, & Benton, 2013). Practices such as planting fruit/seed-bearing plants, limiting the use of pesticides and herbicides, and constructing compost heaps and bird tables increase bird and vertebrate abundance and diversity (Good, 2000). Numerous avian studies have also shown that gardens with sufficient native vegetation can support large populations of both native and exotic bird species at the local level (Daniels & Kirkpatrick, 2006b), and at the landscape level, garden heterogeneity can increase the overall diversity of insectivorous birds (Andersson, Barthel, & Ahrné, 2007). Heterogeneity that includes native plant species may be particularly important, as studies of suburban gardens in Australia show that nectarivorous birds prefer native genera over exotic genera as foraging sites (French, Major, & Hely, 2005).

For non-avian vertebrates, garden size and management style is critical for persistence in urban areas. Baker and Harris (2007) reported 22 mammalian species/species groups recorded in garden visitation surveys within the UK; however, mammal garden use declined as housing became more urbanized (e.g. more impervious habitat) and garden size and structure decreased. Key findings from a range of garden studies show that in addition to high cultivated floral diversity, the three dimensional structure of garden vegetation is an important predictor of vertebrate abundance and diversity (Goddard et al., 2010). Increases in the vegetation structure and genetic diversity of domestic garden habitats have been shown to improve the connectivity of native populations currently limited to remnants (Doody, Sullivan, Meurk, Stewart, & Perkins, 2010) and aid in the conservation of threatened species (Roberts, Ayre, & Whelan, 2007). For example, one study in Latin America documented that garden area and tree height were positively related to the presence and abundance of iguanas within urban areas, and increased patio extent allowed for greater iguana movement across the urban landscape (González-García, Belliure, Gómez-Sal, & Dávila, 2009). In addition to habitat quality, habitat connectivity may also affect the ability of ground dwelling animals to persist in the urban landscape (Braaker et al., 2014); thus, UA systems will need to be connected to other vegetated areas to allow for landscape movement. These studies show that garden structures or management practices that provide food and nesting resources or movement corridors can be important strategies for maintaining vertebrate diversity in cities.

Ecosystem service provision to agriculture and the broader landscape

Ecosystem services are often a function of biodiversity levels (Loreau et al., 2001), thus the composition, diversity, and structure of plant and animal communities within and around UA are important to consider for urban ecosystem service conservation. Specifically, biodiversity provides opportunities for ecosystem services that city planners value–including energy efficiency, stormwater runoff, air pollution removal, carbon storage and sequestration, and water quality provision (McLain et al., 2012). Within agricultural systems, ecosystem services like water storage, pollination, and pest control increase US crop production resilience and protect production values by over $57 billion per year (Losey & Vaughan, 2006; Daily, 1997). However, there remains a large knowledge gap around the provisioning of services in UA systems. This is especially concerning given increasing global food demands, climate-related crop failure, and consistent limitations in fresh food access within urban centers (Thomas, 2010; ver Ploeg et al., 2009).

We posit that UA systems provide a suite of ecosystem services, and that the extent and quality of the services are largely dependent on the biodiversity and vegetative structure of the UA system. Thus the form and management of urban gardens can radically influence service provision. Small garden patches are able to supply structural habitat diversity and carbon storage (Davies, Edmondson, Heinemeyer, Leake, & Gaston, 2011), while allotment gardens can potentially support ecosystem services such as pollination, seed dispersal, and pest regulation to the broader urban landscape (Barthel, Folke, & Colding, 2010). In contrast, reductions in biodiversity can cause a reduction in the resilience of urban ecosystems overall (Colding et al., 2006). Specifically, we review some of the most studied and important ecosystem services to the urban agricultural system: (1) pollination, (2) pest control, and (3) climate control.

Pollination

As mentioned previously, urban agriculture can support a diverse assemblage of bees and butterflies (Matteson & Langellotto, 2010; Matteson et al., 2008; Colding et al., 2006), and the number of native flowering species can positively impact bee abundance and diversity (Matteson et al., 2008; Tommasi et al., 2004). This may have large implications for fruit set and crop production given that crops experience higher or more stabilized fruit set in habitats with greater native bee diversity (reviewed in Winfree & Kremen, 2009). Additionally, floral cover can positively impact conspecific pollen deposition by attracting a greater number of pollinators into an urban garden (Werrell, Langellotto, Morath, & Matteson, 2009).

Some studies suggest that pollinator foraging and dispersal needs are best supported by a network of small, natural habitat fragments across urban areas (Cane 2001). In general, bee foraging distance correlates with body size (Greenleaf, Williams, Winfree, & Kremen, 2007), and some larger bodied bees can regularly fly >1 km from their nest to forage on floral patches (Jha & Kremen, 2013; Greenleaf et al., 2007; Cane, 2001). Thus proximity to natural habitat can increase bee abundance, diversity, and pollination success for a wide range of crop species (Ricketts et al., 2008) and may similarly impact bee diversity within urban landscapes. Research in rural and exurban habitats suggests that bumble bee nesting densities are positively impacted by the proportion of suburban gardens and wooded habitat (Goulson et al., 2010; Jha & Kremen, 2013) and that bees are willing to forage further for high diversity flowering patches (Jha & Kremen, 2013). Furthermore, both nesting density and dispersal are negatively impacted by the amount of impervious cover in a landscape (Jha & Kremen, 2013), revealing the potential for urban landscapes to obstruct pollinator foraging and dispersal. Likewise, heavy development that leads to shaded and closed-off garden areas tend to limit local pollinator diversity (Matteson & Langellotto, 2010). Overall, urban landscapes that maintain diverse natural habitat fragments and minimize impervious cover can promote bee nesting and dispersal. Insights from these studies and others suggest that pollination services may be higher in urban gardens if natural habitat patches and diverse flowering resources are available.

Biocontrol activities/natural pest control

Biological control (or biocontrol) is a method of controlling pest populations through the utilization of other organisms (Flint & Dreistadt, 1998). Biocontrol has been used for centuries within agricultural systems (Flint & Dreistadt, 1998) and could potentially enable sustainable crop production in cities without the reliance of toxic chemical pesticides. This is especially useful in high density urban areas where human exposure to toxins is more risky (Robbins, Polderman, & Birkenholtz, 2001). The natural enemy complex responsible for biocontrol includes predators, parasitoids, and pathogens that regulate pest populations (Fiedler, Landis, & Wratten, 2008). Different natural enemy taxa often have specific habitat preferences; therefore management for biocontrol in urban areas requires knowledge of those factors that influence the abundance and richness of natural enemies.

One of the most effective groups of natural enemies is parasitic Hymenoptera, which reduce herbivorous insect damage to urban trees, ornamental landscape plantings, and residential fruit and vegetable gardens (Isaacs, 2009). Bennett and Gratton (2012) showed that local and landscape scale variables associated with urbanization influence parasitic Hymenoptera abundance and diversity in residential and commercial properties along a rural to urban landscape gradient in Wisconsin. They found that parasitoid abundance was a positive function of flower diversity, and parasitoid diversity decreased as impervious surface increased in the surrounding landscape. This suggests that parasitoids benefit from increased floral resource availability and decreased impervious cover, similar to patterns described for pollinators.

Another effective natural enemy group are the below-ground invertebrates and microbes. Below-ground natural enemies can prey on soil-dwelling stages (eggs, larvae, pupae and adults) of insect pests in urban lawns, often reducing the frequency and intensity of pest outbreaks (Potter, 2005). Yadav et al. (2012) tested if changes in urban habitat structure of gardens and vacant lots influenced below-ground biocontrol services rendered by invertebrate and microbial communities. They showed that ants and microbial communities contributed a majority of the biocontrol service, with ants exhibiting significantly higher biocontrol activity than microbes, particularly in vacant lots. The high levels of belowground biocontrol activity in vacant lots and urban gardens could serve as a foundation for building sustainable pest management practices for urban agriculture in cities.

A number of other natural enemies provide biocontrol services in UA landscapes, such as birds, bats, spiders and beetles (reviewed in Faeth, Warren, Shochat, & Marussich, 2005), but there is still very little research done regarding their role in urban agriculture. The use of organic composts to support pest control (e.g. aphids) by encouraging predatory species (e.g. spiders and beetles) has shown some success (Bell et al., 2008). More work will be required to understand how urban systems, and especially urban agriculture, affect foraging behaviour in higher trophic level natural enemies (Faeth et al., 2005).

Climate regulation

As climate models continue to indicate an increased likelihood of heat waves in urban areas, there has been great interest into the relationship between green infrastructure and mitigation of the urban heat island effect (Alexandri & Jones, 2008). Two main approaches have been proposed as solutions to reduce the urban heat island effect, (1) maintaining more urban green space and (2) reducing impervious surfaces.

Increasing the proportion of green space (e.g. through the development of UA systems) within the urban matrix can reduce both surface and air temperatures (Gill, Handley, Ennos, & Pauleit, 2007). However, the variety of vegetative infrastructure, management, and plant species within UA systems will vary in their cooling potential. Akbari et al. (1997) predicted that up to a quarter of the cooling effect by urban trees in US cities are a result of garden/street trees contributing direct cooling of adjacent buildings, and this effect is dependent on tree size, species, maturity, and architecture. At the garden level, vegetation can influence the energy loads on individual buildings, but how this impacts air temperatures across the wider urban environment is still unclear (Stewart, 2011). However, considering the potential impact that increased vegetation has toward regulating temperatures, there could be big implications on energy use and comfort levels for urban communities. Additionally, gardens located in areas unsuitable for buildings or established as buffer zones along rail corridors and highways, may be helpful in balancing the urban microclimate.

Gardens also provide storm attenuation services to the urban matrix. Vegetation, trees especially, intercept intense precipitation and hold water temporarily within their canopy, thus reducing peak flow and easing demand on storm drains (Xiao & McPherson, 2002). In German cities, allotment gardens used on green belts to facilitate drainage have been shown to reduce heat and demand for air conditioning (Drescher et al., 2006). In contrast, hard paving increases impervious surface, and in Leeds, UK, increased hard paving in residential front gardens has been linked to more frequent and severe local flooding (Perry & Nawaz, 2008). UA systems may contribute to the reduction of impervious surfaces in urban landscapes, thus increasing the drainage and infiltration potential of precipitation.

Potential ecosystem disservices and tradeoffs of UA

As mentioned in the Urban agricultural systems and biodiversity provision section, it is important to understand that not all biodiversity is necessarily "good" biodiversity, and there may be a number of disservices that come from UA that can negatively impact the ecosystem functioning and health of cities. In some cases, there is the possibility of negative types of spill over from managed to natural systems or vice versa of weed, pathogen or pest populations, potentially harming native ecosystems and damaging ecosystem service delivery from natural systems (Blitzer et al., 2012; Zhang, Ricketts, Kremen, Carney, & Swinton, 2007). The juxtaposition of natural systems to UA systems also potentially leads to an increased opportunity for biological invasions and detrimental competition to native species (Niinemets & Penuelas, 2008). Genetic introgression within natural ecosystems by urban garden plants can negatively alter the genetic composition of native vegetative patches and affect the long term viability of these systems (Whelan, Roberts, England, & Ayre, 2006).

UA areas may also lead to increased human health issues and disease transmission to urban populations. For example, UA systems provide increased mosquito breeding sites due to the presence of standing water from irrigation, and this may potentially increase the rate of mosquito-borne diseases in certain areas of the city (Matthys et al., 2010). Additionally, in non-organic UA systems, there is the potential for spillover of chemicals into natural and human habitats, leading to environmental pollution and air- or water-borne health risks (Robbins et al., 2001).

There may also be potential competition for limited resources between UA and natural systems, such as competition for water in arid environments. All such potential disservices from within and outside of UA systems must be considered in optimizing the overall ecosystem services gained in an urban landscape.

A research agenda for promoting biodiversity and ecosystem services in urban agriculture: Challenges and opportunities

There are several research areas that deserve more attention so that urban land use tradeoffs can be better elucidated and biodiversity and ecosystem services (B&ES) can be promoted through the increased integration of UA in cities. We posit that the three key challenges to improving management of gardens for B&ES delivery are (A) availability of urban space, (B) environmental constraints, and (C) lack of knowledge. Below, we provide three major strategies for addressing the key challenges of enhancing B&ES within UA systems.

A. Space availability

Increased urbanization will lead to greater competition for space in cities, making it difficult to maintain biodiversity-supporting habitats. More research is needed to best take advantage of the limited space available for urban gardens and maximize biodiversity within these areas.

Private yards. Private yards make up a significant proportion of green space in a city and do not require the acquisition of new space. Even small-scale, private gardens that present complex vegetation structure can provide appropriate habitats for organisms that have difficulty existing in the urban matrix (Sperling & Lortie, 2010). A number of strategies to incentivize wildlife-friendly gardening activities already exist. In the US, the National Audubon Society's "Audubon at Home" project offers several management principles to increase bird biodiversity in backyards (National Audubon Society, 2013), and the National Wildlife Federation provides certification for 'wildlife-friendly' gardens (National Wildlife Federation, 2013). More research is needed to understand the effectiveness of these incentives to support native biodiversity and food production as many of the techniques are focused on the augmentation of ornamental or floral plants rather than food crops.

Available public spaces. Because greater housing density has been linked to smaller garden sizes, there is an acute need to better understand how UA can be supported within public green spaces, such as community gardens and easements to enhance ecosystem services (Smith, Clayden, & Dunnett, 2009). Even within small habitat strips, the conservation of plants known to attract pollinators or pest natural enemies can provide substantial B&ES while utilizing limited space (Buchmann & Nabhan, 1997), but more information regarding urban plant–animal interactions needs to be known in order to best augment such spaces effectively.

Vacant lots. Vacant lots provide opportunities to create functional green spaces where industrial redevelopment is not likely to happen (Beniston & Lal, 2012). UA in these areas can improve B&ES and provide physical and psychological health for people in cities (Tzoulas et al., 2007). However, a better understanding of how to successfully rehabilitate vacant lots is needed in order to promote this option. Additionally, creating gardens in abandoned lots has implications on urban land tenure for garden management, and it would be helpful to investigate whether temporary gardens make positive contributions to B&ES and food production in the same ways that more permanent gardens do.

B. Environmental constraints

A number of environmental changes come with urbanization and affect the agronomic conditions necessary for food production, such as water availability, nutrient supply, soil degradation, and pest pressure (Eriksen-Hamel & Danso, 2010; Kaye, Groffman, Grimm, Baker, & Pouyat, 2006).

Resilience to climatic change. We need investigations that examine how the choice of garden trees, shrubs, and other plants influence air and surface temperatures in the gardens, and the potential role of garden vegetation to lower energy use and costs in urban environments. Likewise, we need to develop a better understanding of the specific garden plantings that most enhance carbon sequestration in UA. On the flip side, there is basically nothing known about how different UA respond to climate change or climate extremes, and how the urban environment in which UA is embedded may exacerbate climate effects. Thus, more research is needed to understand how plants in UA will respond to increasing temperatures, drought, changes in rainfall amount, nutrient deposition, and weather extremes.

Water use. Research on environmental constraints related to water use is also needed in UA, as irrigation is often required to provide water necessary for urban farming, especially in arid environments. Rainwater or grey water can be used for garden irrigation, and it is cheaper and at times more available than potable water-based irrigation, but UA gardeners must be aware of the potential pathogens and heavy metal contaminants that can cause human and environmental health problems (Qadir et al., 2010), especially with water run-off from these sites.

Soil ecology. Urban soils are usually compacted, have low levels of organic matter, altered soil moisture characteristics, and sometimes have lead or other heavy metal contamination due to urban environmental processes (Beniston & Lal, 2012). A number of methods, such as cover cropping, mulching, producing in raised beds, and changing subsurface drainage through piping, can improve soil conditions to support food production (Beniston & Lal, 2012). However, more research must be done to understand how to sustainably rehabilitate urban soils. The use of both organic and inorganic fertilizers in combination with nutrient-rich wastewater can lead to surplus nutrients in these systems (Graefe, Schlecht, & Buerkert, 2008). Alternative methods, such as "organoponics", where organic compost is used as a growing medium instead of existing soils, need to be further explored to develop farming methods that are successful in the urban environment (Drescher et al., 2006).

C. Lack of knowledge

Two oft-cited barriers to wildlife gardening are the lack of information to alter gardening methodologies for improved biodiversity and the ineffective transfer of knowledge to improve the sustainability of urban gardens (Goddard et al., 2013).

Methodologies for improved B&ES. There is an ever expanding data set on patterns of biodiversity loss in urban areas and the factors that positively correlate with the diversity of plants, arthropods, and vertebrates in urban landscapes. The time is ripe for a quantitative review or meta-analysis of those specific habitat and landscape features of urban habitats (including gardens) that correlate with increases in species richness and abundance of biodiversity in general, and beneficial organisms in particular. For example, past studies document that urban gardens can increase predator or parasitoid diversity (Bennett & Gratton, 2012), but we still lack research that documents the garden features that enhance the specific act of predation by ants, spiders, birds, or other predators. Very little research has focused on how management intensity in UA systems will affect biodiversity and ecosystem services provision from these systems. Additionally, landscape level UA connectivity is important for the creation of networked biodiversity refuges and for the improvement of matrix permeability for organisms. There is a lack of knowledge in understanding how the movement of species between landscape elements can allow organisms to carry out functions at different points in space and time and maintain services that would otherwise be isolated (Lundberg & Moberg, 2003) and how UA fits into the larger general pattern. More research to understand the effects of garden management on landscape-wide biodiversity and movement will be necessary to determine the most critical management practices for promoting effective landscape connectivity.

Knowledge transfer. Increasing our understanding of UA management practices and knowledge spread may be the most important area of research if we are to promote gardens that support B&ES. We need to determine which types of UA contribute disproportionately to food production under different geographic, weather, and socio-economic conditions so that urban gardeners can cultivate specific plant species best suited for their location. Local ecological knowledge is generally low among urban residents; however, discussion between community members may encourage biodiversity-friendly gardening, either through neighborhood or community exchanges of information (Barthel et al., 2010; Colding et al., 2006). More research is essential to understand how to identify the information most useful to urban gardeners and how to most effectively communicate this knowledge.

Conclusion

It is undoubted that UA systems have the ability to increase the biodiversity of urban landscapes. UA systems offer an extensive, varied, and undervalued resource for enhancing urban biodiversity and improving connectivity across the larger landscape. Furthermore, biodiversity supported by UA can increase the quality and quantity of ecosystem services delivered across the urban sphere. However, the increased ecosystem services and disservices that occur with UA must be better understood in order to maximize the benefits for urban society. Based on this review, we suggest that the challenges of limited space, environmental constraints, and knowledge transfer should be further researched to develop methods that incorporate B&ES in urban agricultural systems and identify management methods that maximize B&ES delivery across expanding urban areas.

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