Organic Urban Agriculture

Klaus Lorenz

Abstract: Urban agriculture (UA) has a long tradition in many countries worldwide, is actively engaging about 800 million people, and is now increasingly considered by urban planning and land-use personnel. Urban cropland, in particular, covers more than 67 Mha or more than 5% of the total global cropland area. Urban agriculture practices have many benefits and, in particular, may contribute to food security of urban dwellers by providing vegetables and fruits. However, growing food in urban ecosystems and, especially, on degraded urban soils is challenging, and research on UA in the past has focused on the social sciences. Although the number of studies on urban soils has increased strongly during the last two decades, much work needs to be done as many urban areas have been neglected in previous studies. The needs and benefits of UA and organic agriculture such as building up natural resources through biological mechanism and recycling of wastes, keeping the nutrients cycle within the system, strengthening communities, and improving human capacity are interconnected. Thus, more research is needed on how to maintain or enhance urban soil fertility by soil and land-use management practices. This knowledge must be disseminated among urban gardeners and farmers for improving UA and organic UA systems. Transdisciplinary approaches involving practitioners, urban dwellers, planners, policy makers, and, especially, soil scientists are needed to enhance UA production.

Key Words: Benefits and trade-offs of urban agriculture, urban soil quality, degraded urban soils, soil and land-use management practices

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Urban agriculture (UA) has a long tradition in many cities worldwide, contributing to the lives of many people and providing about 15% to 20% of the world's food (FAO, 2014). However, numerous and disparate definitions of UA exist, making it difficult to estimate its regional and global extent. Several reviews have been published on the global potential of UA in developing and developed countries (Hamilton et al., 2014; Mok et al., 2014). About 800 million people may be engaged actively in UA (Smit et al., 2001). However, uncertainty and accuracy of this previous estimate are not well known (Hamilton et al., 2014). For developed countries, transparent estimates of the global extent of UA have not been published. Otherwise, in developing countries, about 266 million households may be engaged in urban crop production, that is, 29 million households in Africa, 182 million in Asia, 39 million in Latin America, and 15 million in Europe (Hamilton et al., 2014). Thus, UA may have a role in urban food security and self-provisioning especially by producing vegetable and fruit crops (Porter et al., 2014; Zezza and Tasciotti, 2010). Interest in UA has increased substantially in recent years (Grewal and Grewal, 2012), but a better understanding of its global importance for food production is needed. Achievable yields of up to 50 kg m^{-2} $year^{-1}$ have been reported with fruit and vegetable cultivation (Eigenbrod and Gruda, 2015), but estimates of yields in UA are generally not well known (CoDyre et al., 2015).

There are various approaches to urban horticulture such as allotments for self-consumption, large-scale commercial farms, community gardens, and edible landscapes (Eigenbrod and Gruda, 2015). Besides horticultural practices as the main activities, other UA practices are animal husbandry, aquaculture, and arboriculture (Mok et al., 2014). Globally, urban agriculture practices occur at any scale from rooftop gardens to larger cultivated open spaces (Thomaier et al., 2015). For example, the mean farm size in provincial capitals in Poland in the year 2010 ranged between 2.1 ha in Kielce and 23.4 ha in Olsztyn (Kaczmarek, 2014). In addition to contributing to food supplies, the benefits of UA include reduced food transportation distance, carbon (C) sequestration, potentially reduced urban heat island effect, improved physical and mental health, improved aesthetics, community building, employment opportunities, improved local land prices, shortened supply chains, provision of habitat for wildlife, and waste recycling (Mok et al., 2014). For example, greenhouse gas emissions from the food sector are reduced because the amount of food transported from rural agricultural areas is lower (Lee et al., 2015). However, UA can also have negative ecological effects or create ecosystem disservices as gardeners may rely heavily on external inputs, including seeds, plants, water, organic matter, and synthetic fertilizers and pesticides (Taylor and Lovell, 2014; 2015). For example, nitrogen, phosphorus, and potassium may accumulate in high concentrations in soils used for UA. As only some garden waste is composted on-site, much of it enters the municipal waste stream where it contributes to filling of regional landfills and leaves open nutrient cycles within the garden. Furthermore, gardeners import nutrient-rich compost and fertilizers from outside the garden, and a lack of careful management of nutrients may contribute to urban storm water pollution. The unmitigated contamination of UA soils with trace metals and organic contaminants poses a potential threat to the health of gardeners and their families (Taylor and Lovell, 2015). Examples of non–site-related sources of urban soil contamination are bedrock/parent material and dust deposition, especially along traffic routes, whereas site-related sources may be agricultural/horticultural practices and deposits (Meuser, 2010).

Thebo et al. (2014) estimated that around the year 2000, the global area of urban cropland was 67.4 Mha (5.9% of all cropland), with 23.6 Mha irrigated (11.0% of global irrigated cropland) and 43.8 Mha rain fed (4.7% of global rain-fed cropland). Cropland is located within urban extents, especially in Asia. The per-capita area of urban cropland was more than 300 m^2 per capita in developed countries, Commonwealth of Independent States, and South Asia and less than 100 m^2 per capita in Sub-Saharan Africa. The major crops in irrigated urban croplands are rice (*Oryza* spp. L.), wheat (*Triticum* spp. L.), and maize (*Zea mays* L.); wheat, maize, and fodder grasses are the top three rain-fed crops harvested in urban areas. Farmers of urban cropland are producing more rotations per year than farmers overall (Thebo et al., 2014). However, UA has only a limited potential to contribute to global cereal production as the global annual harvested area for cereals is 10 times larger than the global urban area (Martellozzo et al., 2014). In contrast, only one third of the global urban area

Carbon Management and Sequestration Center, School of Environment and Natural Resources, College of Food, Agricultural, and Environmental Sciences, The Ohio State University, Columbus, Ohio, USA.
Address for correspondence: Dr. Klaus Lorenz, School of Environment and Natural Resources, 414B Kottman Hall, 2021 Coffey Rd, Columbus, OH 43210, USA. E-mail: lorenz.59@osu.edu
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would be required to meet the global vegetable consumption of urban dwellers. This global urban area data set aimed to exclude large urban parks but did include some permeable surfaces (e.g., standard backyards; Martellozzo et al., 2014). How much urban area may actually be suitable and available for UA was not considered, however. For example, vegetable yields in urban areas may be lower than rural yields because of naturally low soil fertility and soil degradation. On the other hand, vegetable yields in UA can also be considerably higher than rural yields because of the use of irrigation, relatively high input levels, and the use of best management practices. Most importantly, more science-based information must be generated and distributed among urban farmers to improve the cultivation of plants in urban environments (Wortman and Lovell, 2013). Martellozzo et al. (2014) and Thebo et al. (2014) also highlighted that small urban areas (<100 km^2) with lower population densities can contribute substantially more to UA production than large urban areas. Specifically, small and medium urban areas constitute most of the global urban area, and small urban areas can probably devote a higher proportion of their area to UA because of lower population densities compared with large areas (Martellozzo et al., 2014). However, whereas UA can make a valuable contribution to food security, cities always will depend on a significant external area (Ward et al., 2014).

SOIL-BASED CONSTRAINTS FOR URBAN AGRICULTURE

The last two decades saw a strong increase in the number of studies on urban soils, but soils of many megacities and those of many smaller urban areas have been neglected (Capra et al., 2015). Urban farmers often lack consistent knowledge and skills on methods to preserve urban soil resources (Orsini et al., 2013). Urban soils occur on a continuum, ranging from soils that are undisturbed to those altered by environmental change (e.g., temperature or water regimes) to disturbed soils like those at old industrial sites, building demolition sites, and landfills (Lehmann and Stahr, 2007; Pouyat et al., 2010). The anthropogenic urban soils that have been affected strongly by human activities, such as housing, industrial production, and disposal activities, can be very heterogeneous (Howard and Shuster, 2015). Nevertheless, urban soils provide a limited amount of food in comparison with undisturbed systems and show a moderate potential for enhancement in biomass production for food provision (Morel et al., 2015; Rawlins et al., 2013). Thus, the understanding and management of urban soils are a prerequisite for successful soil-based UA. However, research on UA is concentrated mostly on the social sciences, whereas the natural sciences often are neglected (Pearson et al., 2010; Guitart et al., 2012). This disregard toward the natural sciences is surprising as soil degradation and soil pollution are among the potential crop production constraints in urban areas (Beniston and Lal, 2012; Meuser, 2010). Also, contaminated garden soils pose a risk to human health, but this has not been widely assessed (Taylor and Lovell, 2014). Soils used for UA often are degraded as the use of urban soils to address essential needs such as housing, jobs, and social services takes precedence over UA unless strong land-use and zoning laws are in place (Eriksen-Hamel and Danso, 2010). Thus, land access for urban farmers is difficult, and especially in developing countries, UA often occurs on marginal lands on soils with low fertility on steep slopes, valley bottoms, or in areas adjacent to polluting industries or roads (Orsini et al., 2013).

In the future, UA may be considered increasingly in urban planning and land use as it addresses security, safety, and quality of food. The necessary planner-designer-practitioner dialogue is just beginning in Europe (Bohn and Viljoen, 2014a). In contrast, the American Planning Association's Policy Guide on Community and Regional Food Planning (2007) crossed the divide between food systems planning and urban spatial design. More lawn soils in North American cities may be used in the future for UA as the preferences of home owners are changing (Worrel, 2009). Simply converting a portion of turf grasses to UA in the United States may be sufficient to meet the actual and recommended vegetable consumption by urban dwellers (Martellozzo et al., 2014). Cities such as Vancouver, British Columbia, Canada, and Portland, OR, are focusing on UA research and policies that support small-scale urban farming. Both cities have identified urban lands where food could be grown (Worrel, 2009). Similarly, urban planning reports in cities, such as Detroit, MI, Berlin, Germany, and Leeds, United Kingdom, now recommend the introduction or support of productive urban landscapes including UA (Bohn and Viljoen, 2014a). Recently, the Los Angeles City Council voted to allow Angelenos to plant fruits and vegetables in a strip of city-owned land between the sidewalk and the street (http://www.scpr.org/news/2015/03/04/50192/la-city-council-approves-curbside-planting-of-frui/). Otherwise, a survey of 23 countries in Latin America and the Caribbean indicated that urban and periurban agriculture often is excluded from or not explicitly included in city land-use planning and management (FAO, 2014). Thus, a paradigm shift in urban planning such as zoning land for agriculture is needed to encourage urban and periurban food production.

Urban soil properties can make it difficult to grow a crop because soils may have poor physical properties including poor soil structure, high contents of coarse fragments including technogenic materials like construction waste, high-compaction levels, and impeded water infiltration rates (Bartens et al., 2012). Among soil chemical properties constraining UA are low nutrient contents, contamination especially with heavy metals (e.g., lead), and alkaline pH (Beniston and Lal, 2012). Soil biological properties impeding agronomic production in urban areas include low soil organic carbon (SOC) contents and decreased soil microbial activity. Low SOC stocks of UA soils are particularly worrisome because the multiple benefits of SOC for soil functions and ecosystems services are well known and related to the fundamental role of SOC in the function and fertility of terrestrial ecosystems (Janzen, 2006). However, there is only limited knowledge how specific UA management practices affect SOC contents as well as other major urban soil properties (Edmondson et al., 2014). In contrast to common assumptions, soils used for UA may not be degraded, low in SOC, or compacted. For example, SOC storage was high and soil C:N ratios were low at allotments in Leicester, United Kingdom; and soil quality was consistently high compared with soils from the surrounding agricultural region and compared with English national data (Edmondson et al., 2014). Furthermore, home garden soils in Chicago, IL, had K and P concentrations to 30-cm depth exceeding recommended levels for vegetable gardens (Taylor and Lovell, 2015). The soil organic matter content in those garden soils was also relatively high, averaging 6.4%. Otherwise, soils used for UA at vacant lots in Youngstown, OH, demonstrated high levels of soil compaction (Beniston et al., 2015). Marginal quality of the vacant lot soils for UA also was indicated by their low soil C and soil microbial C concentrations mainly because of removal or mixing/burial of existing topsoil during the demolition activities. Thus, the contribution of UA to higher urban soil quality needs additional research in urban areas worldwide (Edmondson et al., 2014).

ORGANIC URBAN AGRICULTURAL PRACTICES

Many explanations and definitions exist for organic agriculture. For example, the International Federation of Organic Agriculture Movements defines it as "a production system that sustains the health of soils, ecosystems, and people. It relies on ecological processes, biodiversity, and cycles adapted to local conditions rather than the use of inputs with adverse effects. It combines tradition, innovation, and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved." The beginning of organic farming may be traced back to 1924 in Germany with Rudolf Steiner's course on Social Scientific Basis of Agricultural Development (Shi-ming and Sauerborn, 2006). This activity gave birth to biodynamic agriculture, which was developed at the end of the 1920s in Germany, Switzerland, England, Denmark, and the Netherlands. Since then, the research and practice of biodynamic farming, as well as organic, organic-biological, ecological, and natural agriculture, have expanded worldwide (Shi-ming and Sauerborn, 2006). Globally, about 0.9% of the agricultural land was managed in 2014 by organic agricultural practices (Willer et al., 2014). The area of organic agricultural land was 12.2 Mha in Oceania, 11.2 Mha in Europe, 6.8 Mha in Latin America, 3.2 Mha in Asia, 3.0 Mha in North America, and 1.1 Mha in Africa.

Common technologies in organic agriculture to maintain soil fertility and produce high-quality products are (i) applying appropriate rotation programs; (ii) using composts; (iii) using physical, mechanical, and biological mechanisms to control pests; and (iv) applying organic methods in the feed and livestock production (Shi-ming and Sauerborn, 2006). A diverse crop rotation, in particular, is key to crop nutrition and weed, pest, and disease control (Stockdale et al., 2001). Each crop species has slightly different requirements with regard to growing conditions and has different characteristics such as N fixing or N demanding, shallow or deep rooting, and amount and quality of crop residue return. The design of crop rotations allows sequences of crops, which complement and support one another. The inclusion of crops that are able to fix atmospheric N through a symbiotic relationship with N-fixing bacteria that nodulate on crop roots enables organic farming systems to be self-sufficient in N. Examples for mixed intercropping systems are grass/clover (*Trifolium* sp.) leys or cereal with grain legume and cereals intercropped with forage legumes (Stockdale et al., 2001). Among the benefits of organic agriculture are (i) maintaining long-term soil fertility through biological mechanisms; (ii) recycling of wastes of plant and animal origin to return nutrients to the soil, thus minimizing the use of external inputs outside systems, and keeping the nutrient cycle within the system; and (iii) adapting to the local and regional environment conditions and diversified organization (Shi-ming and Sauerborn, 2006). In contrast to organic systems, low-input farming allows the reduced use of certain chemical inputs (Grubinger, 1992).

The use of plant varieties adapted to city conditions and "biointensive" methods, such as intercropping, applying compost, and rotating crops have been recommended to increase yields in UA (Royte, 2015). Urban agriculture depends, in particular, on local and regional human, land, and water resources and products and services in and around an urban area (FAO/WB, 2008). Benefits of UA include water harvesting, water reuse, and urban waste recycling to provide water, animal feed, and fertilizers for demands of UA. Thus, the needs and benefits of organic agriculture and UA such as building up natural resources, strengthening communities, and improving human capacity are interconnected (Iaquinta and Drescher, 2010). Organic UA has, in particular, the potential to reduce the health and environmental risks associated with conventional urban agricultural practices (van Hirtum et al., 2002). Risks by conventional practices may include, for example, contamination of soil, water, and produce by residues of agrochemicals. Certain agricultural practices such as the utilization of biosolids and animal manures, use of agricultural chemicals, management of postharvest residues, irrigation, and tillage operations may be sources as well as facilitators of the transport of nutrients, organic C, heavy metals, pathogens, particulate matters, noxious gases, and pathogens within the environment (Udeigwe et al., 2015). Various health-related issues in humans have been attributed to a number of agricultural pollutants. For example, linkages between cancer and certain agricultural pesticides (Alavanja et al., 2003) and between respiratory diseases and particulate matters ($PM_{2.5}$ and PM_{10}) have been widely documented (Arbex et al., 2007). Likewise, human health issues relating to trace element ingestion (e.g., copper) have been noted (Uriu-Adams and Keen, 2005; Boxall et al., 2009). Pathogens present in animal manure and biosolids have been shown to cause a number of health problems in humans (Mathis et al., 2005; Sidhu and Toze, 2009), whereas environmental contamination by nutrients (e.g., phosphorus) from agricultural sources has also been tied to health risks in humans (Fawell and Nieuwenhuijsen, 2003; Kalantar-Zadeh et al., 2010).

The organic crop production in urban environments is challenging because of intensive plant nutrient requirements and disease incidences (Hernandez et al., 2015). Similar to organic agriculture outside of cities, a major challenge is to find locally adapted plant varieties that will thrive under organic UA conditions (Maddox, 2015). For example, in the absence of chemical supports, organic plant varieties should have strong natural resistance to insects and pathogens and grow quickly and densely to outcompete weeds. Such traits are also the aim of plant breeding for large conventional farms with uniform production inputs, but these conventional varieties are not ideal for the diverse growing conditions found in organic farms (Maddox, 2015). The wide diversity among organic farming systems and among individual farms, in particular, requires a fine-grained adaptation of the crop plants (and animals) used on individual farms (Wolfe et al., 2008). The lack of seeds and varieties suited to OA has been an issue for a long time (Chable et al., 2014). In addition, several current breeding methods do not respect the principles of OA. Participatory plant breeding programs aim to overcome these limitations by initiating collaborations between organic farmers, their organizations, and researchers (Chable et al., 2014). Another approach is using new breeding techniques for rewilding, a process involving the reintroduction of properties from the wild relatives of crops, as a method to close the productivity gap between conventional and organic farming practices (Andersen et al., 2015). Despite the constraints, organic UA is practiced in some cities around the globe and its implementation should be supported by municipalities (Orsini et al., 2013). No estimate on the global extent of organic UA has been published. Furthermore, none of the current organic standards has a specific section on urban horticulture nor is urban production specifically excluded (Schmutz et al., 2014). Thus, Schmutz et al. (2014) recommended how to create a supporting framework for certified organic urban horticulture by allowing certified organic soil–based substrates to be used where it is physically impossible to grow in the soil and the use of novel organic certification systems. In relation to the distance of planting from busy roads, references to organic standards are made in specific recommendations for edible crops in Sweden and for herbs in Austria (Bohn and Viljoen, 2014b). Only a few UA farms are certified as organic. One reason may be that small-scale farmers may be deterred from applying for organic certification given the very complex and costly procedures (Thomaier et al., 2015). Voluntary guidelines may be an alternative approach. For example, Organic Gardening Guidelines, which are based on 60 years of experience by the organization Garden Organic, are widely used in urban areas in the United Kingdom but are not an organic standard (Schmutz et al., 2014).

In Latin American cities, farmers' markets that sell locally grown organic food are spreading (FAO, 2014). Examples are *bioferias* in Quito, Ecuador, and "agrochemical-free vegetable fairs" selling vegetables in Rosario, Argentina. Rosario's vegetables are certified as organic by a system of "social certification," guaranteed by the municipality, the city gardeners' association, Pro-Huerta, and a local NGO that promotes fair trade. Furthermore, the urban and periurban agriculture program in Quito is registered as a producer and marketer of organic produce at the national level. Belo Horizonte municipality in Brazil plans to open a weekly "urban agriculture fair" for direct marketing by farmers who have converted to organic production (FAO, 2014). In Cuba, UA switched to agrochemical-free and labor-intensive practices to increase the food offer, diminish the environmental effects of the Green Revolution production paradigm, and counteract the agro-food crisis caused by the debacle of the socialist block at the beginning of the 1990s (Febles-González et al., 2011). Cuban UA has generally two components, geographical and technological. "Geographical" because these agricultural activities are carried out in or near cities and from very rural to suburban areas and "technological" because practically all the production is agroecological or organic (Koont, 2011). Organic UA is also practiced in China as an increasing number of organic food leisure parks that incorporate agro-tourism into organic UA are constructed (Qiao et al., 2014). For example, the Xiedao Green Resort in Beijing attracts visitors by organic production of many vegetables and on-site treatment and use of wastes from farming and tourism for irrigation and fertilization (Yang et al., 2010). In cities with large supplies of brownfields and vacant lands in the United States such as Baltimore, MD, Detroit, and New Orleans, LA, productive, profitable, organic farms can be established (Vitiello, 2008). Some examples of the effects of organic UA on soil properties in different urban areas and regions are given in the following section.

EFFECTS OF ORGANIC URBAN AGRICULTURE ON SOIL PROPERTIES

Only a limited number of studies on the effects of organic UA on soil properties have been published. In southwestern Nigeria, Africa, organic UA farmers responded to a survey by indicating that they mostly use the practices of minimum tillage, crop rotation, green manure, and sanitation to ensure soil maintenance and fertility (Adebayo and Oladele, 2013). However, soil data were not available. At an organic UA site in Argentina, the application of vermicompost-compost mix and bone meal singly or in combination resulted in an increase in particulate organic C concentration, electrical conductivity, and microbial respiration in 0- to 15-cm depth (González et al., 2010). However, where N and P concentrations of aerial tissue of beet (*Beta vulgaris* L.) were increased by the organic UA treatments, only adding high amounts of vermicompost-compost mix and bone meal in combination (i.e., 2 kg m^{-2} vermicompost-compost mix and 0.15 kg m^{-2} bone meal) resulted in yield increases relative to the control, which received neither vermicompost-compost mix nor bone meal. The productive capacity of soil under urban vegetable production in Rio de Janeiro, Brazil, could be maintained by organic cultivation practices including returning organic matter (OM) with compost, mulching with dry grass, intercropping, and crop rotation (Rego, 2014). The crop rotation was organized according to botanical family, where three species from different families were cultivated, followed by the original species again. Thus, lettuce (Asteraceae) was cultivated, followed by beetroot (Chenopodiaceae), carrot (Apiaceae), and then lettuce again. Again, no soil data were reported.

Organic greenhouse vegetable production on Anthrosols (Inceptisols) in Nanjing City, China, resulted in an increase in pH and OM content to 20-cm depth compared with those for farmland of greenhouse vegetable production (Chen et al., 2014). Soil in the organic system accumulated OM, Pb, and Zn at 0 to 20 cm relative to 80- to 100-cm depth. In comparison with conventional systems, soil Cu and Zn accumulated to 20-cm depth in the organic greenhouse soil. Furthermore, the concentration of soil Cd in the organic system was close to the Chinese environmental quality evaluation standard for farmland or greenhouse vegetable production. Impurities in the applied commercial organic fertilizer that contained large amounts of Cd, Cu, Pb, and Zn were mainly responsible for heavy metal accumulation in the soil under organic greenhouse vegetable production (Chen et al., 2014). In Beijing, China, organic farming development has been recommended by Zhang et al. (2012) to fully use nutrients from human and livestock excreta.

To alleviate soil degradation, *organopónicos* (i.e., raised cultivation beds containing a mixture of about 25% soil and 75% compost) are used in "organic" UA in Cuba (Koont, 2011). Many, although not all, of the *organopónicos*-raised beds are connected to the urban subsoil. No fewer than 10 different crops are maintained at each site per year to benefit from biodiversity (Schmutz et al., 2014). Intercropping systems are also important. The use of organic fertilizers and biofertilizers, microbial and fungal biopesticides, neem (*Azadirachta indica* A. Juss., 1830) products, and worms and worm compost is promoted to manage the *organopónicos*. Mainly manure-based organic fertilizers are used, but the applied amount is often not high enough to maintain soil fertility. Thus, Körner et al. (2008) recently emphasized the potential for integrating composting the organic fraction of municipal solid waste to increase the amount of production. Plant growth promoters isolated from vermicompost and applied directly to lettuce (*Lactuca sativa* L.) leaves promoted yields under organic lettuce production (Hernandez et al., 2015). However, there are no standards for Cuban urban horticulture with independent third-party certification as for organic farming. Nevertheless, some enterprises are certified as organic farms by international certification bodies, but the vast majority of UA sites in Cuba are not (Kilcher, 2009; Schmutz et al., 2014).

As indicated previously, the objectives of organic UA may be undermined by inputs of heavy metals and other pollutants such as atmospheric deposition of heavy metals from industrial and urban sources (Säumel et al., 2012). For example, Pandey and Pandey (2009) reported that heavy metals accumulated in 0- to 20-cm depth at organic farming sites in the urban agglomeration of the city of Udaipur, India. The increased heavy metal contents in the soil altered soil porosity, bulk density, water-holding capacity, microbial C, carbon substrate–induced respiration, alkaline phosphatase, and fluorescein diacetate hydrolytic activities. The impaired quality of soil and of substrates available for decomposition in the heavy metal–contaminated soil may have resulted in a reduced rate of decomposition and, thus, contributed to altered chemical and physical soil properties. Based on a comparison of open organic farming plots with those in a glasshouse, Pandey and Pandey (2009) concluded that the atmospheric deposition was the main contributor to raised heavy metal contents in edible plant parts, especially in fruits and leaves.

Allotment gardens on natural soil in Lisbon, Portugal, had higher soil organic matter contents to 30-cm depth under organic than those under conventional practices (Cameira et al., 2014). However, high N inputs mainly from organic amendments (i.e., biocomposts of green waste and food waste and/or animal manures) resulted in high mineral N contents and nitrate losses by leaching. Thus, the organic production system was per se not environmentally safer with respect to nitrate pollution than the conventional production system (Cameira et al., 2014). Through long experience, households practicing some type of organic urban horticulture in Matara City, Sri Lanka, have recognized the importance of composted manure in adding nutrients and improving soil structure for growing their horticultural plants (De Zoysa, 2007). However, most of the households faced difficulties in finding organic fertilizer and were highly concerned about the availability of compost-making technology as an important opportunity to promote urban horticulture.

IMPROVING URBAN SOILS BY ORGANIC AGRICULTURE

Similar to understanding the effects of conventional UA practices on soil quality (Edmondson et al., 2014), effects of specific organic UA practices on soil properties are poorly known. For UA practices in general, Beniston and Lal (2012) have summarized intensive management strategies to reduce soil-based constraints to production requiring few external energy inputs, having low costs, and using locally available resources. In the following section, some soil-improving UA practices will be discussed and the potential to increase SOC stocks by organic UA.

An improvement in urban soil properties may be possible by adding substantial inputs of OM and nutrients with manure, biomass composted on-site, and commercial composts. Those practices may be reasons why soils of allotments in Leicester, United Kingdom, had 32% higher SOC concentrations and 36% higher C:N ratios in 0- to 7-cm and 7- to 14-cm depth than pasture and arable soils and 25% higher TN and 10% lower bulk density values than arable soils (Edmondson et al., 2014). However, it was unclear whether the addition of burned biomass including tree, shrub, or hedge trimmings, sweet corn (*Zea mays* var. *rugosa* Bonaf.) stalks, and brassica (*Brassica* spp. L.) roots, together with diseased plants and noxious weeds, contributed to the improvement in soil properties. Furthermore, about one half of the studied plots received synthetic fertilizers, and this action may have also improved allotment soil properties in Leicester (Edmondson et al., 2014). The importance of OM amendments for improving properties of a physically degraded urban soil was highlighted by a study on vacant lots in Youngstown (Beniston et al., 2015). Two years after applying large quantities (150 Mg ha^{-1}) of compost produced from urban yard waste, soil physical properties (i.e., bulk density, % water-stable aggregates, mean weight diameter, and total porosity) to 10-cm depth were improved compared with the unamended control but not physical properties in the 10- to 20-cm depth. However, plant-available nutrient and microbial biomass C pools increased at both depths after applying OM amendments. Cover cropping with sorghum-sudangrass (*Sorghum bicolor* × *S. bicolor* var. *sudanese* cv. BMR) was also a recommended practice for the degraded urban soil as it increased crop yields and improved soil physical properties by producing large quantities of cover crop biomass (Beniston et al., 2015).

Maintaining SOC stocks (i.e., Mg SOC ha^{-1}) and, in particular, SOC flows (i.e., Mg SOC ha^{-1} year^{-1}) aids agricultural production (Janzen, 2015). However, the action of any farming practice is to export C from a site. Thus, to maintain or increase soil fertility under organic UA, SOC stocks and flows should either be maintained or increased, specifically, when degraded urban soils are converted for organic UA use. Recommended practices may include, for example, reduced tillage, improving plant nutrition, adding organic amendments, and irrigation (Janzen, 2015). Abundant organic amendments in urban areas can be compost and biochar. Specifically, large amounts of organic urban waste are produced annually, and some of it may serve as feedstock for biochar production (Renforth et al., 2011). Adding biochar during the management of organic UA soils may enhance the SOC stock particularly. For example, studies by Beesley and Dickinson (2011), Ghosh et al. (2012), and Scharenbroch et al. (2013) indicated that adding biochar to urban soils improves soil quality and enhances SOC stocks. However, research on the use of biochar is in its early stages and the application of biochar to urban soils and gardens needs additional work (Renforth et al., 2011; Lorenz and Lal, 2014). Nevertheless, biochar is used already on urban farm fields and city lawns (Cernansky, 2015).

According to a recent meta-analysis, organic farming practices have the potential to accumulate SOC (Gattinger et al., 2012). Important for this increase may be C inputs with organic fertilizer, mainly in the form of manure, slurry, or compost. Thus, to improve soil properties and agronomic productivity for organic UA, SOC should be maintained or increased by adding OM amendments as discussed previously. However, less well studied are effects of organic farming practices on SOC stocks in tropical regions and also those in subsoil horizons below 15-cm depth (Gattinger et al., 2012). Crop rotations are probably also important for increasing topsoil SOC stocks under organic farming practices. Furthermore, multicropping and crop rotations were associated with an average yield suppression of 9% and of 8%, respectively, for organic compared with conventional practices, whereas organic yields were on average about 19% lower than conventional yields (Ponisio et al., 2015). Thus, besides adding organic amendments, diverse crop rotations should be among management strategies for organic UA. However, it is unlikely that the yields of conventional UA can be achieved by organic UA practices. Specifically, maximizing organic UA yields probably can be achieved only by adding synthetic fertilizers, an action that is not an organic farming practice (Edmondson et al., 2014; Gattinger et al., 2012). Adding N to conventionally managed soils results in higher nitrous oxide (N_2O) emissions than from organically managed soils when based on the area of cultivated land but lower emissions when based on crop yields (Skinner et al., 2014). For equalizing the N_2O emissions per yield, a yield increase in the organic systems of 9% would be necessary, and crop rotations show some potential to close this yield gap (Ponisio et al., 2015). Furthermore, organic agriculture may lower methane (CH_4) emissions from soils because of lower mineral N contents in the soil solution, resulting in less suppression of the activity of enzymes for microbial CH_4 oxidation compared with soils under conventional management (Skinner et al., 2014). However, conclusions are preliminary as the meta-analysis on N_2O and CH_4 emissions by Skinner et al. (2014) was based on only 12 studies that cover annual measurements and all were conducted in temperate regions of the Northern Hemisphere.

FUTURE RESEARCH AND DEVELOPMENT FOR URBAN AGRICULTURE

Food production in urban and periurban areas may contribute to "city region food systems" and has the potential to alleviate urban food insecurity (FAO/WB, 2008). The production of fresh perishable food should especially be promoted because this practice has comparative advantages to rural agriculture. However, to assess the importance of UA, more production data are needed for urban areas differing in size worldwide (Martellozzo et al., 2014). Most importantly, UA should be integrated fully into development planning. However, awareness among policy makers must be created about the risks and opportunities of UA and the need to integrate it in agriculture development strategies, national food and nutrition programs, and urban planning and resource management. This integration includes a proper understanding of local conditions in relation to the need for urban development. For example, whereas soil-based UA depends on soil fertility, urban soils also fulfill a range of other ecosystem functions and services (e.g., flood regulation, pollution attenuation, regulation of biodiversity, carrying structures and piped utilities; Rawlins et al., 2013; Morel et al., 2015), which should be integrated toward the sustainable development of urban areas. Interdisciplinary, multisectoral, and participatory (i.e., transdisciplinary) approaches are needed to find sustainable solutions for UA. A close collaboration between communities, waste and water departments, urban planners, and health authorities, but also private companies and other stakeholders, is needed. Specifically, the private sector must be directly involved in planning decisions, and experiences should be shared with local decision makers and actors from the public and private sectors, including NGOs, and growers' representatives (FAO/WB, 2008).

As discussed previously, UA is now considered increasingly by urban land-use planning. Similar to creating green zones for parks, botanical gardens, and golf courses within city boundaries, zones for UA should be equally integrated into urban planning (FAO/WB, 2008). A legal and institutionally simple framework for UA must be created, accompanied by an adequate institutional and operational framework for the implementation and monitoring of the policy, because that framework defines to a large extent the efficiency of the policy (RUAF-Foundation, 2002). Comprehensive policies are needed to regulate urban development and protect prime soils for UA. Thus, necessary land-use conversions within urban areas should be directed toward areas with urban soils less suitable for UA (Yigini et al., 2012). Otherwise, suitable vacant urban spaces should be made productive by UA practices (Mougeot, 2006). However, most producers have no secured tenure status, which precludes any substantial investment in terms of soil fertility or infrastructure. Temporary occupancy permits for urban producers should be tested as a strategy to get (temporary) access to land, especially for the poor urban producers (Mougeot, 2006). Furthermore, UA often is not supervised and exposed to the "innocent" use of pesticides and polluted water. Thus, the lack of agricultural extension services provided to the urban producers must be addressed (FAO/WB, 2008).

The competition for basic resources (water, soil) between UA and other priority urban needs (e.g., drinking, domestic and industrial water use; infrastructure construction) must be addressed (FAO/WB, 2008). Research is needed on the contribution of UA to maintain the quantity and quality of urban natural resources. To ease the competition for water, the use of recycled treated wastewater should be promoted, and decentralized water treatment facilities and low-cost technologies must be developed. The use of wastewater in UA also would supply some nutrients required for agricultural crop production. Furthermore, the water resource management (e.g., irrigation, drainage facilities) must be adapted to climate change. In addition to land, soil, and water, labor is another principal resource in competition for other uses than UA (FAO/WB, 2008). Thus, data on the economic sustainability of UA are needed. For example, the high costs of "urban inputs" may to some extent be compensated for by the better prices obtained at the farm gate and the short marketing chain as compared with rural agriculture (FAO/WB, 2008).

More research is needed to cope with the biophysical challenges of growing crops in urban soils. Awareness must be raised among stakeholders toward the recognition of urban soil properties and the spatial distribution of soils of different qualities within an urban area (Yigini et al., 2012). Enhanced recognition is among the prerequisites for best management of urban soils for UA. As discussed previously, urban soils may be contaminated, particularly with heavy metals, and risk assessments are needed to establish safety of a site before its use for UA (Sharma et al., 2015). To improve degraded soils for UA, huge quantities of organic waste products are available in urban areas. The application of organic wastes as compost or biochar to improve biological, chemical, and physical properties of soils for UA must be studied in urban areas worldwide (Edmondson et al., 2014). This investigation should also be accompanied by an assessment of closing nutrient cycles by urban waste recycling as a contribution to the sustainable development of urban areas (Taylor and Lovell, 2015). Science-based knowledge on the risks associated with soil-based UA and on the improvement of soils for UA must be distributed among urban producers. More studies on the potential for organic UA of soils of different urban regions worldwide are needed. A critical assessment is needed of whether conventional or organic UA is more economically and socially sustainable by making the best use of urban soils and other principal urban resources.

CONCLUSIONS

The importance of UA may increase in the future as urban population and rural-urban migration are increasing. Urban agriculture can provide important contributions to the creation and maintenance of multifunctional urban landscapes and, in particular, to the provision of fruit and vegetables to urban dwellers. Thus, UA can improve food supply, health conditions, local economy, and social integration and support sustainable resilient urban development. The benefits of UA are interconnected with organic agriculture. However, studies on UA and on organic UA generally neglect natural and soil science, although understanding and management of urban soils are a prerequisite for maintaining soil fertility. In particular, soil degradation and atmospheric deposition of pollutants may limit horticultural production in urban areas. Thus, the constraints to crop production in urban areas and how those can be reduced need additional research. Based on this improved knowledge, urban producers need to be educated on recommended soil and land-use management practices for improving (organic) UA systems. Transdisciplinary approaches involving practitioners, urban dwellers, planners, policy makers, and, especially, soil scientists are needed to enhance (organic) UA production.

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