Considerations for reducing food system energy demand while scaling up urban agriculture

Environmental Research Letters

LETTER • OPEN ACCESS

Considerations for reducing food system energy demand while scaling up urban agriculture

To cite this article: Eugene Mohareb *et al* 2017 *Environ. Res. Lett.* **12** 125004

Environmental Research Letters

LETTER

OPEN ACCESS

RECEIVED
15 September 2016

REVISED
21 August 2017

ACCEPTED FOR PUBLICATION
25 August 2017

PUBLISHED
5 December 2017

Considerations for reducing food system energy demand while scaling up urban agriculture

Eugene Mohareb[1,7], Martin Heller[2], Paige Novak[3], Benjamin Goldstein[4,5], Xavier Fonoll[6] and Lutgarde Raskin[6]

1 School of the Built Environment, University of Reading, Reading, United Kingdom
2 Center for Sustainable Systems, University of Michigan, MI, United States of America
3 Department of Civil, Environmental, and Geo-Engineering, University of Minnesota, MN, United States of America
4 Division of Quantitative Sustainability Assessment, Technical University of Denmark, Denmark
5 School for Environment and Sustainability, University of Michigan, MI, United States of America
6 Civil and Environmental Engineering, University of Michigan, MI, United States of America
7 Author to whom any correspondence should be addressed.

E-mail: e.mohareb@reading.ac.uk

Keywords: urban agriculture, energy, food systems, resource efficiency, industrial ecology, local food, food–energy–water nexus

Abstract

There is an increasing global interest in scaling up urban agriculture (UA) in its various forms, from private gardens to sophisticated commercial operations. Much of this interest is in the spirit of environmental protection, with reduced waste and transportation energy highlighted as some of the proposed benefits of UA; however, explicit consideration of energy and resource requirements needs to be made in order to realize these anticipated environmental benefits. A literature review is undertaken here to provide new insight into the energy implications of scaling up UA in cities in high-income countries, considering UA classification, direct/indirect energy pressures, and interactions with other components of the food–energy–water nexus. This is followed by an exploration of ways in which these cities can plan for the exploitation of waste flows for resource-efficient UA.

Given that it is estimated that the food system contributes nearly 15% of total US energy demand, optimization of resource use in food production, distribution, consumption, and waste systems may have a significant energy impact. There are limited data available that quantify resource demand implications directly associated with UA systems, highlighting that the literature is not yet sufficiently robust to make universal claims on benefits. This letter explores energy demand from conventional resource inputs, various production systems, water/energy trade-offs, alternative irrigation, packaging materials, and transportation/supply chains to shed light on UA-focused research needs.

By analyzing data and cases from the existing literature, we propose that gains in energy efficiency could be realized through the co-location of UA operations with waste streams (e.g. heat, CO_2, greywater, wastewater, compost), potentially increasing yields and offsetting life cycle energy demands relative to conventional approaches. This begs a number of energy-focused UA research questions that explore the opportunities for integrating the variety of UA structures and technologies, so that they are better able to exploit these urban waste flows and achieve whole-system reductions in energy demand. Any planning approach to implement these must, as always, assess how context will influence the viability and value added from the promotion of UA.

Introduction

Urban agriculture (UA) has been undergoing a global resurgence in recent decades, with cities in both advanced and emerging economies implementing programs to encourage its use (Mok *et al* 2013, Orsini *et al* 2013, Hamilton *et al* 2013, Vitiello and Brinkley 2013). This renewed interest has led to the exploration of the extent to which UA could be expanded, including a number of investigations that estimate the potential for UA to meet local food demand; for example, Grewal and Grewal (2012), McClintock *et al* (2013) and Goldstein *et al* (2017), suggest provision of total food demand (former) and vegetable demand (latter two), of 4.2%–17.7%, 5% and 32%, respectively. Expanding UA is expected to improve local sustainability, including benefits to social (addressing food deserts, building community cohesion, or higher intake of fresh produce) and economic (cash crop production, reduced food costs) facets of cities. The environmental aspects associated with the net direct and indirect energy implications of UA will be the primary sustainability focus area of this research.

Part of the rationale for reconsidering UA has been its potential environmental benefits, including reductions in energy demand throughout the food supply chain. As a result, UA has been included in greenhouse gas (GHG) mitigation strategies for cities (Arup and C40 Cities 2014) and broader urban sustainability agendas through multi-city agreements and partnerships, such as the UK's Sustainable Food Cities Network and the Milan Urban Food Policy Pact, the latter of which includes 100 large cities around the world (Milan 2015, Andrews *et al* 2017). However, when considering the complex interplay between food production, energy requirements, and water availability (i.e. the food–energy–water nexus), the ability of UA to reduce energy demand is unclear.

This review article examines energy use in the food system, explores the opportunities that exist for high-income cities to increase the energy/resource efficiency of this overall system through UA, and proposes changes that could be made in the planning of cities to enable greater reductions in energy demand, with a focus on the United States. The scope extends beyond the frequently-assessed topic of transportation into topics such as embodied energy of production inputs (i.e. water, nutrients, heating, CO_2), reduction in packaging, storage, and processing needs. This review aims to provide a point of reference for energy considerations that should be made if UA is going to provide a greater share of the global food supply.

Classifying urban agriculture

Estimating the current scale of UA is difficult and varies based on how it is defined; for example, Thebo *et al* (2014) estimate that there were 67 megahectares (Mha; 10^6 ha) of UA[8] globally in 2000 (5% of global arable land in that year; Food and Agriculture Organization 2010, table A4), with roughly 1/3 of the UA area being irrigated. Their quantification includes spatial data where agricultural areas and urban boundaries with populations greater than 50 000 overlap, most of which would be classified as peri-urban[9] agriculture and would not capture small-scale operations such as residential gardens, vacant lots, or building-integrated production (e.g. balcony gardens, rooftop gardens). Inclusion of peri-urban agriculture would produce a substantially higher estimate of UA than the area that is currently used in these more commonly-perceived forms of UA. Looking at the scale of some of these types of UA, Taylor and Lovell (2012) examine the total area of UA in the city of Chicago using 2010 aerial photographs. They find that approximately 0.04% of Chicago's land area of 606 km^2 was being used for urban agriculture; of this, nearly half (45%) was in residential gardens, while most of the remainder was in vacant lots (27%) and community food gardens (21%). To provide a sense of scale of the opportunity to expand urban agriculture, a 2000 study of vacant land in US cities finds that those in the Midwest had an average of 12% vacant land, and a national average of 15% (Pagano and Bowman 2000)[10].

As alluded to above, UA manifests itself in a number of different structures and locations within the built environment. Attempts have been made in the literature to classify UA; Mok *et al* (2013) identify three distinct scales of agriculture in urban systems. These are (in order of decreasing size): small commercial farms and community-supported agriculture, community gardens, and backyard gardens. All of these UA scales differ in their structure, inputs, and productivity; as a result, their net impact on life cycle energy demand, and other resource inputs, also varies. Goldstein *et al* (2016b) further classify UA to consider structure and inputs in a taxonomic scheme, based on the conditioning required for the growing environment (temperature, light and CO_2 control) and integration within the surrounding urban system (building integrated or ground based). They claim that both features are important to UA energy regimes, with space conditioning (particularly the need for heating in cold climates) being an essential consideration, along with the potential for building integrated farms to utilize dissipative heat and CO_2 to offset production inputs.

A broad classification of UA is provided in table 1, which is roughly ordered by scale and sophistication of production. It should be highlighted that while the preservation of peri-urban agriculture can be captured in assessments of UA, the focus of this review is on approaches to scaling up UA that

[8] Thebo *et al* (2014) define urban agriculture as the spatial coincidence of agricultural areas with urban extents with populations over 50 000.

[9] Peri-urban agriculture refers to agricultural production that occurs at the urban–rural interface.

[10] Data include vacant land with and without abandoned buildings; Chicago did not provide data for this study to allow a direct comparison, hence the average area for Midwest cities is provided here; as well, it is not being suggested here that all vacant land be allocated to, or are suitable for, UA.

Table 1. Type of urban agriculture associated with structure/location of production, potential beneficial energy impacts relative to intensive rural agriculture, and requirements for upscaling.

Type of urban agriculture	Authors' definition	Potential direct energy benefits	Considerations for successful upscaling	Sources
Residential gardens	Open air or protected[11] food production occurring within the boundaries of a residential property, primarily for personal consumption	• Non-mechanized inputs •Reduced cold chain/retail requirement (onsite end-consumption)	• Knowledge dissemination for production, preservation • Regulations for application of fertilizers, pesticides • Appropriate crop selection	(Kulak *et al* 2013, Altieri *et al* 1999)
Allotment and community gardens[12]	Open air or protected food production occurring upon community or municipally-owned land, primarily for personal consumption	• Non-mechanized inputs • Reduced cold chain/retail requirement	• Municipal allocation of green space • Expedited application approval to facilitate utility connection • Mulch from municipal greenspace maintenance	(Leach 1975)
Rooftop/balcony agriculture	Open air or protected food production occurring on structures built for other primary functions, for either personal consumption or commercial availability.	• Thermal transfer from rooftop • Improved yield • Improved building insulation • Onsite waste diversion	• Building code consideration (structural, utilities)	(Sanyé-Mengual *et al* 2015, Saiz *et al* 2006, Specht *et al* 2013, Grard *et al* 2015, Orsini *et al* 2014)
Industry/ residence-integrated greenhouse	Controlled-environment food production with supplemental heating, integrated into structures built for other primary functions that involve purpose-built infrastructure for yield improvement towards commercial availability.	• Waste heat/CO_2 utilization • Improved yield	• Inventory of urban resource streams • Zoning by-laws to enable co-location of agriculture with resources	(Zhang *et al* 2013)
Vertical farms	Controlled-environment food production with supplemental heating, in multi-story structures developed with the primary function of crop production for commercial availability. Generally located within urban boundaries	• Onsite waste diversion (e.g. waste-to-feed for livestock operations) • Potential for on-site nutrient cycling • Improved yield	• Building code changes (structural, utilities) • Innovations in lighting, agriculture system integration in built environment • Low-carbon grid due to expected substantial energy requirements	(Despommier 2013, Hamm 2015)
Peri-urban agriculture	Open air, protected, or supplemental heat environment food production at the urban-rural interface. Generally for commercial availability, but may include subsistence agriculture in developing-world contexts.	• Preservation of high-yielding prime agricultural land	• Legal protection of peripheral farmlands from incompatible urban development	(Francis *et al* 2012, Krannich 2006)

are integrated into the built environment, rather than on maintaining existing agricultural land in the urban periphery. Hence, large scale conventional peri-urban agriculture is beyond the scope of inquiry here.

[11] Protected food production refers to enclosed environments (e.g. with polyethylene or glass) that are not climate-controlled; controlled-environment food production includes both protected environments and those with supplemental heat.

[12] Urban or peri-urban agricultural space designated and protected by municipalities or community groups for non-commercial purposes.

Energy consumption in the food system and urban agriculture

The modern food system encompasses a broad collection of energy end-users. Starting from the agricultural phase through transportation of food to retailers and households, and culminating in waste handling, the current predominantly linear structure of the food system is highly dependent on energy inputs for its operations of production, processing, distribution, consumption and disposal of food products (Pimentel *et al* 2008). Examining the US case, the USDA ERS

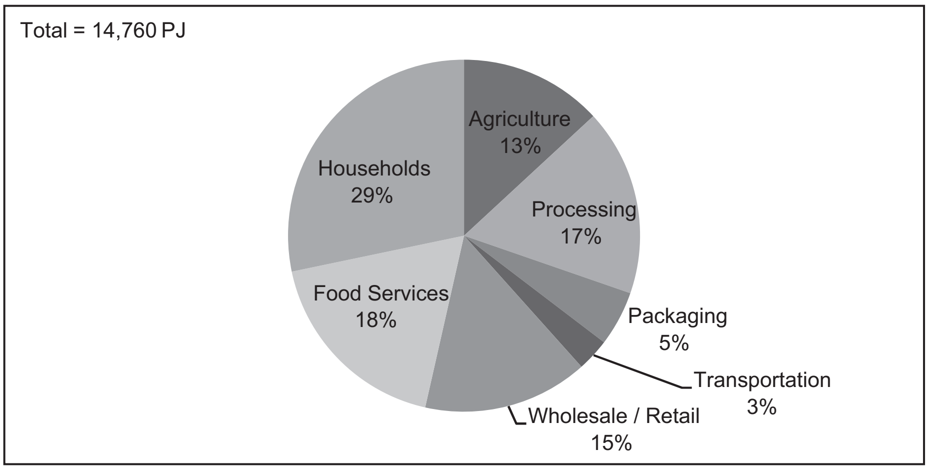

Figure 1. Energy consumption in the US food system in 2002 (adapted from Pelletier *et al* 2011, from Canning *et al* 2010).

Table 2. Energy and water demand per unit yielded for various tomato production systems (modified from Goldstein *et al* 2016a).

Production system	Irrigation water (m Mg^{-3})	Direct and indirect energy demand (MJ Mg^{-1})
Ground-based non-conditioned (two cases)	50; 74	6500; 2600
Ground-based conditioned	65	33 000
Building-integrated non-conditioned	68	3300
Building-integrated conditioned	9	56 000
Conventional (conditioned)	2	10 000

(2010) estimates that nearly 14.4% of total national energy consumption in 2002 was food-related. A breakdown of this consumption is provided in figure 1.

The majority of energy use in the food system occurs beyond the farm gate; the United Nations Food and Agriculture Organization (FAO) estimates that over 75% of energy use in the food system of high-income nations occurs after cultivation (Food and Agriculture Organization 2013). This is consistent with the 2002 US analysis in figure 1, which suggests that the post-agricultural energy use share is over 87%. However, the potential for UA to impact energy demand beyond production is substantial (e.g. packaging, processing, transportation, waste management), as discussed below. In addition, figure 1 excludes wastewater and food waste treatment; therefore, a complete consideration of energy use associated with the expansion of UA will require an examination of not only food production but also energy inputs across the entire food system, including waste handling and treatment. Changes in energy use relative to the status quo must also investigate the food–energy–water nexus to validate the environmental case for scaling up UA and avoid any unintended shift of impacts from one resource system (i.e. energy) to another (i.e. water).

Energy benefits of urban agriculture

Proponents suggest a number of energy-related benefits are realized through the reintroduction of food production within cities (Howe and Wheeler 1999, Garnett 1997, Smit and Nasr 1992, Kulak *et al* 2013). Studies most commonly highlight savings in transportation energy, reduced storage requirements at the wholesale/resale level, and energy inputs of food waste/loss along the supply chain, but also include additional biomass provision from silviculture (i.e. to offset energy imports; Smit and Nasr 1992), easier exploitation of resource use (Zhang *et al* 2013), and lower resource-intensity of production (Kulak *et al* 2013). Meanwhile, peri-urban agriculture can preserve higher-yielding prime agricultural land (Krannich 2006, Francis *et al* 2012), which has the potential to provide less resource-intensive production. Looking at more sophisticated integrated operations (vertical farms, integrated greenhouses), exploited waste streams (CO_2, heat, macronutrients) could offset energy requirements that are required for providing these inputs in conventional operations (Despommier 2013, Zhang *et al* 2013). Additionally, if the distributed nature of UA can be supported by a similarly distributed energy infrastructure system, food/agriculture waste can be digested locally to generate biogas for heat or electricity production, further decreasing the energy footprint of UA. Energy-related benefits associated with the various structures/locations of UA have also been described in table 1 (excluding transportation).

Interactions with other components of the urban food–energy–water nexus

Urban agriculture has the potential to affect energy-related components of the food–energy–water system within urban boundaries and beyond. Suggestions of positive and negative impacts, both within and beyond the urban boundary, are presented in table 2.

It is important to note that energy demand for services required in UA can differ from those provided through open-field agriculture. An exploration of literature that can provide greater insight on how these different UA approaches can influence energy needs follows.

Energy demand for UA water systems

Energy demand in irrigation systems are a noteworthy component of scaled-up UA that must be considered in order to avoid inadvertently increasing demand relative to conventional open-field systems. Irrigation systems in an open-field agricultural setting are relatively low-energy when compared with potable urban water systems that could be used in UA; in one study open-field irrigation energy demand is estimated at 0.63 MJ m^{-3} water (Esengun *et al* 2007, used in the absence of a similar US case study). However, in a UA system, potable water may be used for irrigation and generally requires substantially more energy for treatment, with the Electric Power Research Institute (2002) suggesting an estimate of 1.3 MJ m^{-3} and 1.7 MJ m^{-3} for public utilities using surface and groundwater, respectively (including distribution), for a hypothetical 10 million gallon per day treatment plant. Meanwhile, Racoviceanu *et al* (2007) estimate energy demand at 2.3–2.5 MJ m^{-3} treated water used in the City of Toronto's water treatment. The Racoviceanu *et al* (2007) study considers a surface water source, and includes chemical fabrication/transportation, treatment, and onsite pumping, though most of total energy intensity (~70%) is attributable to untreated and treated water pumping. Data on Massachusetts' 2007 energy demand for water treatment and distribution suggests an average value of 1.4 MJ m^{-3} (US Environmental Protection Agency 2008), whereas California's 2005 report on the energy-water relationship provides estimates of 1.4 MJ m^{-3} and 9.7 MJ m^{-3} for Northern and Southern California, respectively (range attributable to differences in energy required for conveyance from source to treatment facilities; Klein *et al* 2005). This latter California report also suggests that when desalination options are employed in water treatment, an additional 9.3–15.7 MJ m^{-3} and 3.7–9.3 MJ m^{-3} are required for seawater and brackish groundwater, respectively. It is worth noting that depth of groundwater source, pumping requirements for surface/groundwater, and on-farm treatment will influence the energy demand and could bring this figure closer in line with that from water utilities.

The types of secondary energy used can also vary for different types of irrigation, influencing both cost, overall energy efficiency, and GHG emissions. For example, Ontario, Canada's field crop irrigation is typically powered by diesel systems, while greenhouse irrigation is generally powered by electricity (Carol 2010). Diesel has an emissions intensity of 74 kg $CO_2e\ GJ^{-1}$, while electricity grid GHG intensity in Ontario was 14 kg $CO_2e\ GJ^{-1}$ in 2014 (IPCC 2006, chapter 3). For comparison, US electricity emissions intensities ranged from 1 to 266 kg $CO_2e\ GJ^{-1}$ in 2012 (US EPA 2015).

Water/energy trade-offs for UA production methods

Water use can be mitigated through the use of more water-efficient growing systems (such as hydroponic systems), though these can result in increased energy demand in pumping and lighting, and associated GHG emissions. For example, hydroponic[13] systems have been shown to have lower water demand than soil-based production, in addition to avoiding the need for a solid growing medium and the associated energy inputs of its provision (Albaho *et al* 2008). However, Barbosa *et al* (2015) have modeled energy and water demand for hydroponic and conventional production systems for lettuce; while water demand was reduced by 92% (250 to 20 l $kg^{-1}\ y^{-1}$), energy demand increased by 8100% (1100 to 90 000 kJ $kg^{-1}\ y^{-1}$), due primarily to heating and cooling loads (74 000 kJ $kg^{-1}\ y^{-1}$), artificial lighting (15 000 kJ $kg^{-1}\ y^{-1}$) and circulating pumps (640 kJ $kg^{-1}\ y^{-1}$).

Focusing on energy, Shiina *et al* (2011) study hydroponic urban 'plant factories' (temperature controlled, artificial lighting and humidity controlled) in Japan, and show that the energy intensity of the production resulted in estimated greenhouse emissions of 6.4 kg $CO_2e\ kg^{-1}$ lettuce, despite the operation's high yields. Continuing to use GHG emissions as a proxy for energy demand, this compares with estimates of 0.2 and 0.9 kg $CO_2e\ kg^{-1}$ for lettuce from Michigan hoop houses and California open-field lettuce production (Plawecki *et al* 2014), and ranges between 0.24–2.62 kg $CO_2e\ kg^{-1}$ for lettuce from European open field and hothouse production (Hospido *et al* 2009). Meanwhile, Goldstein *et al* (2016a) compared cumulative energy demand of rooftop hydroponic greenhouse tomatoes and 'conventional' production and find the former to be roughly ten times as energy intensive, with important implications for carbon footprint. However, switching energy source from the Massachusetts electricity grid to hydroelectric or solar PV makes rooftop hydroponic greenhouse production less carbon intensive than conventional production.

These demonstrate that are potential for trade-offs when addressing environmental footprints through UA if focusing on a single performance metric (i.e. water alone). Though, as hydroponic growing systems can be used in controlled, protected, and open-field growing systems and with a wide selection of hydroponic technology options available, variation can be expected in the yields and energy demand of hydroponic operations; this introduces uncertainty in applying these

[13] Hydroponic systems are those that involve the culture of plants in the absence of soil in a nutrient-supplemented water medium ('Hydroponics', in Anonymous 2017).

figures to specific contexts, but underscores the need for careful consideration in designing for energy and water demand reduction.

Alternative irrigation sources

Urban agricultural systems provide an application for rainwater collection, as well as black/greywater[14], all of which could reduce wastewater volumes and stormwater runoff, and potentially improve surface water quality and decrease net energy use as a result (i.e. due to the avoidance of UA irrigation with potable water and downstream wastewater treatment). As examples, wastewater treatment in California and Massachusetts is estimated to require, on average, 1.7 and 2.4 MJ m^{-3}, respectively (US Environmental Protection Agency 2008, Klein *et al* 2005). This has the potential to be reduced if conveyance and treatment requirements are avoided through application of wastewater in UA. Further, if stormwater can be diverted from treatment plants to UA in jurisdictions using combined sewer systems, energy demand, as well as pollutants to receiving bodies, could be reduced. In an extreme case, substantial diversion of rainwater for UA from lakes and rivers that ordinarily receive it could contribute to local/regional ecosystem decline or surface water quality issues (Goldstein *et al* 2016a). Finally, depending on how UA is managed, runoff from open field urban farms could result in increased nutrient loads being passed down to receiving bodies or downstream wastewater treatment plants (Pataki *et al* 2011). Upscaling UA could result in this being an additional source of non-point pollution for consideration by city managers/planners.

Packaging materials

The use of packaging materials can also potentially be avoided in UA operations, in instances of production for personal consumption or within shorter distribution chains such as when food is sold directly by the producer (Garnett 1999). For example, the climate impacts of the embodied energy of polyethylene terephthalate clamshells and polystyrene trays that are often used in tomato packaging (again, using carbon as a proxy for energy use) were estimated to be 25% and 100% greater, respectively, per unit mass of tomato when compared to loose packaging (US Environmental Protection Agency 2010). Still, the authors noted that modified atmosphere packaging using plastics have been shown to increase shelf life by two or three times, which may reduce waste and, consequently, GHGs associated with tomato production and disposal. This waste reduction could then offset the embodied energy needed for the packaging material that provides this added shelf life.

[14] Blackwater refers to wastewater conveying faeces and urine, while greywater includes other wastewater streams from human use that do not (i.e. dishwater, shower water).

The use of packaging does not need to be an all or nothing proposition; employing some packaging for various meal components can result in a net energy savings (relative to 'typical' packaging configurations) when accounting for avoided waste and marginal energy requirements; semi-prepared meals examined by Hanssen *et al* (2017) were slightly more energy efficient when compared with those prepared from scratch. It is generally important to recognize the embodied energy of the food products and packaging materials being considered; higher embodied energy food products (cheese, beef, bread) more easily justifying the additional energy inputs associated with packaging than unprocessed fruits and vegetables (Williams and Wikstrom 2011). Similarly, the application of plastic films and containers may be more easily justified when compared with more energy-intensive materials such as steel, aluminum, or glass.

Transportation and supply chain considerations

While UA and other forms of localization are often intuitively thought to reduce life cycle energy demand, the reality is more complicated (Webb *et al* 2013). Supply chains crossing a variety of artificial jurisdictional boundaries may in fact be more direct than those created by constraining agriculture within a region/state, depending on the product, consumption point, and regional characteristics (Nicholson *et al* 2015). Broad-scale localization of agriculture has the potential to increase transportation energy, as well as associated GHG emissions, relative to the conventional supply chain if definitions of local and implications for modified supply networks, including transport modes, are not carefully considered. Indeed, a commonly cited reason to pursue UA is to reduce energy-related impacts associated with transportation. Estimates of transportation's contribution to the food system's energy demand and GHG emissions have been estimated at approximately 10% or less (Weber and Matthews 2008, USDA ERS 2010, Garnett 2011).

Numerous studies from the literature (Coley *et al* 2009, Edwards-Jones *et al* 2008, Pirog *et al* 2001) have challenged the common assumption that 'localizing' food production results in reduced transport energy use and GHG emissions, and effects on distribution networks need to be evaluated on a case basis to justify such a claim. For instance, transport-related impacts for cheese shipped 20 000 km from New Zealand to consumers in England by boat were dominated by road-freight and consumer automobile use, highlighting the limitations of singular focus on transport distance (Basset-Mens *et al* 2007). The GHG implications of external energy inputs to support year-round urban food production and their ability to overwhelm gains achieved through reduced distribution distances must be considered in the context of upscaling of urban food production.

Urban heat island mitigation

The predominance of dark (low-albedo) surfaces in cities results in the absorption of solar radiation and elevated temperatures in and around urban areas, raising the demand for cooling energy (the urban heat island effect; Oke 1973). Urban agriculture could play a role in attenuating this phenomenon, by increasing surface albedo and the cooling effect of plant evapotranspiration (Ackerman *et al* 2014). Vegetation situated on buildings has been shown to reduce individual building cooling demands in Toronto, Canada, Madrid, Spain and La Rochelle, France (Bass and Baskaran 2001, Saiz *et al* 2006, Jaffal *et al* 2012). Ackermann and colleagues estimated that scaling up UA in New York City could reduce the local urban heat island by 22%–44% (~1 °C), mitigating energy demands for cooling (Ackerman 2012). The importance of this ancillary benefit of UA could become more important with the increasing frequency and severity of heat waves under likely climate change scenarios (Jansson 2013).

Impact of type of production system

Assuming UA may involve the use of protective structures or controlled environments, it is relevant to consider the energy demand associated with such structures. Generally speaking, open-field and protected agriculture (e.g. hoop houses with no supplemental heating) have been found to require lower energy inputs than heated systems (e.g. heated greenhouses). Studies focusing on open-field conventional tomato production in the US and the Mediterranean had energy inputs for production of 140–280 MJ Mg^{-1} (Brodt *et al* 2013, Tamburini *et al* 2015). An average of three Moroccan protected tomato operations had energy inputs of diesel and electricity for fertigation and pesticide application of 460 MJ Mg^{-1} (Payen *et al* 2015). With hothouse operations, energy input can increase further, with a selection of studies focusing on tomato cultivation showing energy inputs ranging from 425, 28 500, 76 000 MJ Mg^{-1} for case studies in Northern Italy, France, and Iran, respectively (Heidari and Omid 2011, Boulard *et al* 2011, Almeida *et al* 2014). In the French case, heated operations required six times more energy per unit of weight than the protected system (Boulard *et al* 2011). Goldstein *et al* (2016a) found similar patterns of variation for tomatoes depending on production method, with resource requirements presented in table 2 (modified here to present consistent units).

Nevertheless, studies that directly compare controlled-environment growing with open-field agriculture for certain crop types present a mixed picture. In one study, Martínez-Blanco *et al* (2011) found that life cycle cumulative energy inputs per Mg of protective structure greenhouse tomatoes produced in Catalonia was 13% greater when compared with open-field production (considering operations using mineral fertilizer inputs only). The additional energy demand in the greenhouse operations is dominated by the greenhouse structure, in spite of some savings realized through reduced cultivation-stage fertigation infrastructure, nursery plants and irrigation needs. However, in an Indonsian case study, Kuswardhani *et al* (2013), found that energy demand per unit mass was higher for open-field tomato when compared to protective structure greenhouses, but lower for lettuce; this is attributed to higher fertilizer and pesticide/herbicide needs for open-field tomatoes (predominantly the latter), whereas open-field lettuce had lower energy requirements in spite of this higher demand (and higher labor inputs) due to the substantial electricity requirements for the drip irrigation system used in the greenhouse lettuce. Their study did not include the embodied energy of the greenhouse structure.

Studies for tomato production in Antalya, Turkey suggest that energy requirements per kg yielded for protective structure greenhouse tomato production were approximately 30% lower than that in open fields (Esengun *et al* 2007, Hatirli *et al* 2006). The greater yield coupled with lower labor, machinery, and irrigation energy provide a net energy saving relative to open fields, in spite of greater fertilizer, electricity, and pesticide inputs for these greenhouses. This study also excludes embodied energy of greenhouse infrastructure. When taken together, these studies suggest that inputs required for UA will be operation, crop, and climate dependent, emphasizing the need for consideration of these elements when making comparisons and considering UA expansion.

With respect to soilless production systems, Albaho *et al* (2008) state that aeroponic[15] systems require an uninterrupted electrical supply but it is unclear as to whether this energy demand is offset by lower inputs and higher yields relative to conventional controlled-environment or hydroponic systems. A summary of the energy implications of production methods is provided in table 3, along with estimates of energy implications from efforts to scale up UA in table 4.

Drivers of variability

Judging the pressures production systems have on resource demands requires reflection on a number of contextual factors. For example, local climate/geography may reduce the need for energy-intensive inputs (i.e. mild climate, plentiful surface/rain water). As well, existing infrastructure (green and grey) may or may not provide access to necessary inputs (nutrients, water, energy, labor, and growing media). This reflection may also include questions such as whether there is an abundance of low-grade heat that is accessible for exploitation and is the supplier (i.e. a local utility) amenable to supporting its exploitation, or perhaps if there is an existing agreement to

[15] Aeroponic systems are those that involve the culture of plants in the absence of soil or hydroponic media (Anonymous 2011).

Table 3. Energy implications of different production methods.

Production method	Energy benefits	Energy costs
Open air—large scale	Reliant on natural systems for photosynthesis, growing environment, and, to some extent, water supply	Centralized and seasonal production systems that tend to require complex distribution networks that necessitate transportation and cold storage
Open air—small scale (e.g. balcony, allotment, residential garden)	Reliant on natural systems for photosynthesis; avoids conventional distribution network	Input practices dependent on skill of UA practitioner (potential for excessive use), system design (e.g. moisture retention of planter boxes compared with field)
Controlled environment—protected agriculture	Higher yields; can be located close to consumption, with an extended growing season; low material inputs relative to other	Relatively high embodied energy inputs of capital per production unit when compared with open field
Controlled environment—conventional greenhouses	Higher yields; can be located close to consumption, with an extended growing season	As above, but with energy inputs for lighting, irrigation systems, or other control systems, in addition to growing medium
Controlled environment—advanced soilless systems	Higher yields; can be located close to consumption, with an extended growing season	As above, but with added operating energy from soilless systems (e.g. pumping, dosing equipment)

Table 4. Estimated energy impacts within and beyond urban boundaries from scaling up urban agriculture on the broader food–energy–water system.

Within urban boundaries	Beyond urban boundaries
Upward Pressure • Heating (for some controlled environment agriculture) • Water/wastewater treatment (conventional network usage) • Labor (paid or unpaid) • Transportation (in cases of inefficient local supply chain)	**Upward Pressure** • Construction materials (e.g. steel framing, LDPE sheeting, polycarbonate glazing)[a,b,c]
Downward Pressure • Transportation (e.g. backyard gardens) • Waste disposal (assuming less loss along supply chain) • Water/wastewater (decentralized usage) • Building energy demand (e.g. evapotranspiration, green roofs)	**Downward Pressure** • Irrigation water (through controlled-environment agriculture) • Inorganic inputs (wastewater reuse) • Machinery/capital (human inputs) • Packaging materials • Cold-chain requirements

[a] Goldstein *et al* (2016a).
[b] Martínez-Blanco *et al* (2011).
[c] Kulak *et al* (2013).

supply nutrients from wastewater to peri-urban agriculture or further afield. Additionally, an abundance of uncontaminated vacant land or a low population density may make open-field or protected systems the most plausible approach. Further considerations with respect to publically-owned land might be whether these local green spaces are compatible with UA integration, when safety, waste collection, accessibility, and public demand are taken into account. Finally, Pelletier *et al* (2011) suggest that scale of production systems may also play a role in energy efficiency; though scale in itself is not an indicator of energy efficient production, smaller operations have been observed to have lower energy intensities in the examples of tomatoes and swine. It is clear that further research is needed to parse out the roles that scale, climate, existing infrastructure, waste resource availability can have on the overall energy picture of UA operations. Moreover, an assessment of the local context is necessary before promoting any particular UA approach, along with the accompanying resource demands these systems require in a given context.

Exploiting urban resources for local agriculture

Numerous opportunities exist to scale up UA in an energy-efficient manner, both within present urban systems and carefully-planned future developments. If, however, an industrial ecology lens were applied for future planning, a paradigm shift in food systems integration could be achieved with respect to the urban food–energy–water system, including opportunities for utilizing food waste, wastewater, and waste heat/CO_2 recovery. In industrial ecology, efforts are made to mimic natural ecosystems through more efficient use of resources through the exploitation of waste streams by other production systems (Clift and Druckman 2016).

The urban form can be re-imagined to facilitate the incorporation of UA in a truly integrated way. The concept of co-locating agriculture would imply more than preserving peri-urban agriculture and household gardens; it would focus on identifying spaces within built-up areas that are amenable to agriculture and that are also within close proximity to agricultural inputs (waste heat, compost, wastewater and flue CO_2 from compatible sources). One example of such an eco-industrial system in a rural setting is described by Zhang *et al* (2013), where yields can be improved from CO_2 fertilization through the integration of manure management and greenhouse operations. Biogas generated from the manure disposal system is used in place of natural gas to heat the greenhouses and fertilize with CO_2, while reducing emissions of GHGs and air pollutants. Metson *et al* (2012) demonstrate that the co-location of agriculture near urban areas can enable improved resource efficiency. In their Arizona study, they found that the increasing dairy demand from a growing city was accompanied by an expansion of dairies and alfalfa farms (for feed) in its hinterlands; the alfalfa farms utilized cow manure from the dairies, as well as biosolids from urban wastewater, as a source of phosphorous, increasing the local nutrient cycling in the city-region. If planners are able to identify or (ideally) inventory projected/current UA-related resource streams, the overall embodied or direct energy demand associated with these UA systems can be reduced more deliberately and, presumably, more effectively.

A summary of key resource streams that are valuable in agriculture is provided in table 5, along with their conventional energy inputs as stated in a variety of literature sources. The extent to which these energy demands will be offset will differ depending on the agriculture operation.

With the increasing frequency of extreme weather events and uncertainty of future water availability, agriculture production in the US has the potential to be negatively affected by climate change (US Global Change Research Program 2014). Urban agriculture could increase resilience against these (as it historically has done during resource shocks through the centuries, per Barthel and Isendahl 2013), while reducing environmental impacts within the current infrastructural construct; these benefits could be even greater if an industrial ecology approach is taken. Indeed, controlled-environment production systems can potentially protect crops from the climate variability and extremes that would otherwise disturb open-field production systems. These more secure, and higher yielding (Martínez-Blanco *et al* 2011) operations would bring greater certainty in yields as well as improved resilience, relative to the uncertainty of the broader food supply chain. In addition, controlled-environment agriculture systems can be planned for integration into new and existing buildings and industries, to make better use of inputs that are predominantly from urban waste streams (e.g. flue gas, waste heat, wastewater, biosolids). The following sections provide a discussion of strategies to deploy controlled-environment agriculture within the current infrastructural context and within an interconnected UA ecosystem that is designed for resource recovery from waste streams.

Energy production from food waste

Food waste has the potential to be converted to a useful energy resource in the form of biogas, with many cities already collecting source-separated organics for processing in local anaerobic digesters (Uçkun Kiran *et al* 2014, Sanscartier *et al* 2012, Mohareb *et al* 2011, Bernstad and la Cour Jansen 2011). Following the potential for circular resource use suggested by Metson *et al* (2012), the proximity of increased urban food waste from both production as well as further down the food supply chain could provide a greater feedstock for co-located urban anaerobic digestion (AD) systems. In addition, digestate produced from these facilities could find local end-uses in UA operations, facilitating a circular material flow. Governments are currently promoting UA to reduce the carbon footprint of cities (Arup and C40 Cities 2014). Keeping this objective in mind, it is important to consider how food waste (a major component of GHG emissions from landfills; US EPA 2017) can be better utilized within a more cyclical UA system.

Using food waste for energy generation through AD provides an opportunity for distributed energy generation while decreasing the impact of food waste on downstream systems (landfills, wastewater treatment plants). Levis and Barlaz (2011) assessed the environmental performance of food waste disposal in nine common waste management systems and found that AD performed best with respect to GHG emissions, NO_x, SO_2 and net energy demand. Further, considering the proximity to potential end users, the use of biogas from AD facilities for both heat and electricity production could become more economically attractive in an urban context, especially with local UA consumers of waste CO_2 (from biogas production) and AD digestate. It is estimated that the US cities produce 130 Mt of food waste annually[16]. Using estimates of 184 kWh of electricity and 810 MJ heat Mg^{-1} of wet waste (from Møller *et al* 2009), this quantity of food waste has the potential to provide electricity for 7.2 million Nissan Leaf all-electric vehicles[17] and the equivalent heating demand for over 1.5 million Michigan homes[18], respectively.

Cities are currently operating AD facilities that are providing energy to the broader community. Barcelona is treating 192 000 t yr^{-1} of its organic fraction of municipal solid waste (OFMSW) through AD, having a positive energy balance of around 2.2 MJ produced/MJ consumed at the facility from pre-treatments and digester pumping/stirring (Romero-Guiza *et al* 2014).

[16] Uses an estimate of 500 kg of food discarded per capita in 2010 from retail and consumers (USDA ERS 2013) and a US urban population of 261 427 500 (US Census Bureau 2015).

[17] Assuming 11 500 miles per year (Heller and Keoleian 2015), Leaf mileage of 29 kWh/100 miles (www.fueleconomy.gov/).

[18] The average Michigan home consumes 123 million BTU, 55% for heating (www.eia.gov/consumption/residential/reports/2009/state_briefs/pdf/mi.pdf).

Table 5. Key agricultural resource streams, potential urban sources, and energy requirement for resource stream use in conventional urban agricultural systems.

Urban resource stream	Potential alternative urban sources	Energy requirement–conventional sources	Source of energy requirement data
Treated water	• Decentralized wastewater treatment • Rain barrels • Grey water	1.33–1.40 MJ m^{-3} (surface water) ~1.73 MJ m^{-3} (groundwater)	Electric Power Research Institute (2002)
Heat and carbon dioxide[a]	• Electricity generation • Residential furnaces, boilers, hot water heaters • Industrial/commercial waste heat • Anaerobic digesters • Heat transferred from conditions buildings • Sewage networks	~2500 kWh m^{-2}-year (mild climate; e.g. HDD18 = 2800 Abbotsford, BC[e], greenhouse heated with natural gas)	Calculated from British Columbia case study (Zhang *et al* 2013)
Nitrogen		13.8 MJ kg^{-1} (34.5% NH_4NO_3) 14.5 MJ kg^{-1} (NH_4SO_4) 15.1 MJ kg^{-1} (27.5% NH_4NO_3) 32.58 MJ kg^{-1} (CH_4N_2O)[c]	Audsley *et al* (1997), Danish and UK data
		EU average–35.28 MJ kg^{-1} (urea); best–1.84 MJ kg^{-1}	Smith *et al* (2001)
		57.46 MJ kg^{-1} (US)	West and Marland (2002)
		Feedstock–25.52–27.65 MJ kg^{-1} (UK) indirect and direct energy—8.4–19.6 MJ kg^{-1} (UK)	Mortimer *et al* (2003)—NH_4NO_3; appendix C
Phosphorus	• Digestate from anaerobic digestion • Human biosolids • Animal manure • Compost (i.e. using wastes from gardens, green roofs, and UA) • Industrial waste streams	3.82 MJ kg^{-1}	Hansen (2006)[b]
		9.72–18.72 MJ kg^{-1} (EU)	Audsley *et al* (1997)
		EU average–36.22 MJ kg^{-1}; best–1.82 MJ kg^{-1} (P_2O_5)	Smith *et al* (2001)
		7.02 MJ kg^{-1} (P_2O_5) (US)	West and Marland (2002)
		15.80 MJ kg^{-1} (P_2O_5) (EU)	Elsayed *et al* (2003)
Potassium		0.54 MJ kg^{-1}	Hansen (2006)[b]
		5.00 MJ kg^{-1} d	Audsley *et al* (1997)
		EU average–11.20 MJ kg^{-1}; best–0.58 MJ kg^{-1} (K_2O)	Smith *et al* (2001)
		6.84 MJ kg^{-1} (K_2O) (US)	West and Marland (2002)
		9.29 MJ kg^{-1} (K_2O) (EU)	Elsayed *et al* (2003)
Calcium		1.73 MJ kg^{-1} ($CaCO_3$) (US)	West and Marland (2002)
		2.09 MJ kg^{-1} (CaO) (EU)	Elsayed *et al* (2003)
Structural materials	• Municipal solid waste for construction materials (e.g. hoop houses)	0.11 MJ kg^{-1} steel (for hoop house or greenhouse structures)	Althaus (2003) - EcoInvent 3, Life Cycle Inventories of Metals 2009

[a] to be diverted to boost yields of greenhouse operations.
[b] excludes 'inherent' (embodied) energy of CH_4, 30.5 MJ kg^{-1} N.
[c] including mining energy demand, as reported in Bøckman *et al* 1990.
[d] sum of natural gas, electricity and coke used in manufacture of chromium steel.
[e] five-year average (2012–16) from www.degreedays.net.

Additionally, anaerobic co-digestion with sewage sludge could enhance biogas production and deals with the seasonality that food waste from UA can present (Fonoll *et al* 2015, Shrestha *et al* 2017). Policy interventions will likely be necessary to encourage broader investment in AD (Binkley *et al* 2013). For example, in the north of Italy, 26 000–28 000 of OFMSW are treated each year in AD plant; while the facility has obtained a positive cash flow of €2.5 million yr^{-1}, an incentive for the use/generation of renewable energy was needed to enable this to occur (Riva *et al* 2014).

Beyond energy production, AD offers additional benefits. Situating anaerobic digesters near UA operations could facilitate the reuse of digestate (such as in Garfí *et al* 2011), saving on fertilizer requirements and reducing transportation costs for waste diversion. The coupling of AD with pyrolysis has the potential to produce biochar, which could be used to improve soil fertility (Monlau *et al* 2016). Excess heat from AD or pyrolysis can also be applied to the digester to or to district heating systems and can be used to heat houses or aquaculture operations.

The barriers associated with the reintroduction of livestock into relatively dense areas are formidable; these include local regulations, public health concerns, and logistic difficulties of feed provision (Food and Agriculture Organization 2001, Butler 2011). If surmounted, these operations, as well as primary and secondary food processing industries (e.g. breweries, ethanol production, harvest-related waste from agricultural operations) can provide substantial feedstocks for AD.

Finally, in cases where AD is impractical, UA provides a local end user for composted residues. Hence, onsite compost facilities could be a component of future UA operations. This would reduce GHG emissions from waste that would have been disposed of in a landfill, and avoids the need for transportation of waste to a location offsite. According to the US EPA WARM model[19], composting food waste and avoiding its addition to landfill results in a net reduction of 0.96 Mg CO_2e per Mg of food waste.

Wastewater reuse in urban agriculture

Both solid and liquid streams of wastewater are an underutilized resource, with their current perception as a municipal liability requiring resource-intensive treatment and disposal. It has been estimated that approximately 2% of the total US electricity use is for municipal wastewater treatment (Electric Power Research Institute 2002). The aeration step of treatment, which promotes biodegradation of pollutants, accounts for approximately 50% of this energy use (Curtis 2010, Mamais *et al* 2015). This approach also results in the release of GHG emissions to the atmosphere; in 2000, US wastewater treatment resulted in ~33.3 Mt CO_2e from energy use and sludge degradation (Center for Sustainable Systems 2014). A system that diverts wastewater from treatment, reduces the level of treatment, or eliminates the need for aeration (through diversion from receiving water bodies to UA) could help reduce these emissions.

Wastewater reuse could be a practical source of water and nutrients in UA. Previous studies have noted heavy metal and pathogen contamination of wastewater-irrigated produce (Amoah *et al* 2007, Khan *et al* 2008), underscoring the need to ensure regulatory requirements for irrigation water quality are met (World Health Organization 2006). If cities/neighborhoods were to reorient their wastewater treatment goals from a focus on disposal to one of reuse, the treatment reduction could result in substantial energy savings—directly at the point of treatment, as well as upstream from crop production. For example, crops grown using water and nutrients recovered from wastewater could offset the embodied energy demand of crops that are grown elsewhere using more energy-intensive irrigation water and inorganic fertilizers. Anaerobic membrane bioreactors are one technology that has been proposed to accomplish these goals (Smith *et al* 2012, 2014), recovering energy, generating an effluent rich in nutrients and low in suspended solids and organics, and eliminating energy requirements related to aerobic treatment (Smith *et al* 2014). Regardless of the technology used, further research is necessary to evaluate the removal potential of trace contaminants and viral pathogens prior to reuse for UA (Smith *et al* 2012, McCurry *et al* 2014). By taking an industrial ecology approach, residential waste streams and industrial waste streams that are relatively benign and with a low pathogen load (e.g. brewery waste) could be used in subsurface irrigation of UA crops, avoiding conventional treatment and reclaiming nutrients for food production.

Waste heat or CO_2 use for urban agriculture

Finally, a further industrial ecological approach would see conventional infrastructure systems integrated with agriculture to increase productivity. Many sources of waste heat and CO_2 exist within the urban boundary, from residences to industrial operations to electrical utilities. Where natural gas is employed in these applications, greenhouse operations can utilize the relatively clean exhausted low-grade energy as a heat source, as well as CO_2 for crop fertilization (Kimball 1983, Mortensen 1987). If greenhouses and households could be integrated, there is a potential efficiency gain in the combined system over its discrete components, including through the provision of CO_2 for crop fertilization and utilization of waste heat. A number of studies have suggested that building-integrated agriculture has the potential to improve overall energy performance of the system (Specht *et al* 2013). Decentralized residential heating systems in single-family homes make utilization challenging, but specialized building-integrated systems, like the example developed by Seawater Greenhouses, could be a model for smaller-scale units that utilize waste heat and CO_2 on site (Delor 2011). Nevertheless, the model presented by Cerón-Palma *et al* (2012) of a rooftop greenhouse in Barcelona highlights the challenges of building-integrated UA, as greenhouse heating requirements were not temporally aligned with the times of excess heat within the building; instead, this type of production system may be better suited to colder climates where exhaust CO_2 and heat from boilers/furnaces are more available during winter months. This highlights the need for additional research on how to overcome these types of management issues to support greater resource efficiency.

[19] Using national average landfill characteristics and default waste hauling distances of 20 miles (www3.epa.gov/warm/).

Planning and human capital considerations for urban agriculture

Historically, UA was a natural part of urban development and, eventually, an essential component of the plans of early urban planning practitioners (Vitiello and Brinkley 2013). However, UA was not a primary objective for planning developed-world public spaces in industrialized food system of 20th century cities. Calls to reconsider the value of UA have been made for decades (e.g. in the pattern language proposed by Alexander *et al* 1977) and planning for UA, as a result, has returned. The success of UA re-adoption in urban design is demonstrated by the Carrot City Initiative (Gorgolewski *et al* 2017) which facilitates discussions on urban design for food production. These and other resources can help to increase the sophistication of food planning in a more cyclical urban ecosystem.

Planners can open up or create space to enable the upscaling of UA in either building-integrated systems or new/existing green space. For example, parks could be redeveloped from being merely aesthetically-pleasing recreational landscapes to be more functional, with edible productivity through the incorporation of fruit trees and community gardens. Inventories of suitable public and private vacant land could be identified for UA use through geomatic methods (McClintock *et al* 2013). Municipal support for training in the harvest and processing of crops could increase the public's awareness of the resources embodied within the food they consume and minimize and potentially minimize crop waste. Processing infrastructure, such as fruit presses or preserving facilities, could be situated within the park's borders. By-laws could be put in place to incentivize rooftop UA, as has been done with green roofs in some cities (e.g. Toronto and Chicago; Loder 2014)

As mentioned previously, UA expansion could lead to local increases in polluted run-off. This may require the implementation of by-laws restricting fertilizer or pesticide application, storm water remediation/mitigation measures, and out-reach to inform citizens of health and environmental implications of agriculture. As well, inventories of UA and surveys of practices coupled with geographic information systems could help planners identify potential hotspots for runoff, odors, or other impacts.

Human labor is an abundant urban resource that is anticipated to become more available in cities as trends of urbanization and automation progress. Smaller-scale agricultural systems have the potential to utilize this labor, as they tend to be more labor intensive than conventional mechanized open-field agriculture. As well, the integration of UA in buildings and the application of advanced production approaches (i.e. soilless operations) require specialized training during design, construction and operation, creating high-skilled employment opportunities. The impacts on food prices by shifting to small-scale UA systems is unclear; the 2012 US agricultural census suggests that hired and contract farm labor contributed to only 10.2% of total farm production expenses, though it is suggested that this would vary substantially by crop raised and potentially less mechanized/automated systems (US Department of Agriculture 2014, USDA ERS 2014). The recreational utility realized by those pursuing UA as a leisure activity could reduce the net increase in costs (i.e. people providing free labor in pursuit of UA as a hobby); further, multiple non-monetary benefits (civic engagement, social cohesion, food security) have been recognized, enabling a scenario where broad public benefits of UA can be realized, coupled with an understanding of its effects on health and the environment (Chen 2012, Horst *et al* 2017).

Avoiding unintended consequences in scaling up urban agriculture

A number of issues may inhibit efforts to scale up UA, including land scarcity (Martellozzo *et al* 2014), UA's uncertain contribution to food security (Ward 2015), environmental impacts of decentralized production (Nicholson *et al* 2015, Coley *et al* 2009), and management of new sources of food waste (Levis and Barlaz 2011, Forkes 2007, Smil 2004). Avoiding unintended consequences and continued inefficiency in the food system through urban production requires a planning approach that coordinates input streams, reduces potential for waste and enables co-location to mitigate growth in transportation demand. Foley *et al* (2011) suggest that efforts to meet the food needs of the rising global (urban) population face substantial challenges to environmental protection. Further, resource demands of all urban food consumption far exceeds the resources that can be provided within city boundaries and moving towards this goal could create new local resource stresses; for example, Ramaswami *et al* (2017) demonstrate this situation for New Delhi's water demand, where water used for food production represented 72% of urban-related withdrawals (in turn, only 14% of these water withdrawals was provided within the city's boundary).

We argue that an industrial ecological approach to UA has the potential to slow land use change (through the intensification of production), increase crops yields (by increasing management intensity), increase resource efficiency (through co-location of inputs from waste streams), and encourage low-carbon diets (through increased access to fresh produce; Wakefield *et al* 2007, Schafft *et al* 2009). However, proximity alone are not a guarantee for success of eco-industrial UA; Gibbs and Deutz (2007) review a number of unsuccessful industrial ecological case studies and interview participants in these and find that results often do not match objectives. However, with an incremental planning approach, improved networking to develop trust and cooperation, and targeted policy interventions by municipalities could improve the success of industrial ecological approaches.

Implications of UA on production inputs, food waste and transportation (of both labor and food products) are dependent on UA approaches taken. As an illustration, this will be influenced by the production practices of UA practitioners, efficiency of distribution systems, public and active transportation options for accessing UA sites, producer and retail practices for food disposal, and local attitudes towards food waste. All of these require further study within each local context.

Conclusions

This review has examined UA through a novel lens, considering the energy implications of promoting the expansion of food production in various forms within cities in advanced economies. Scaling up UA has implications for the broader energy system, with the potential to affect direct and upstream energy demand, and enable the utilization of resources to a greater degree. This review underscores the need to pursue further case study research to understand the implications of human and physical geographies on net energy demands and other environmental impacts of UA in its many iterations. Different combinations of crop type, climate, production method/scale, availability of 'waste' resources, co-location approaches, and intensity of production all need to be explored to obtain a broader understanding of the life cycle energy implications of scaling up urban agriculture.

We have proposed and provide supporting information for a resource-efficient path to pursuing the expansion of UA—through the exploitation of crop and other food wastes, reuse of municipal wastewater and biosolids for crop fertilization and irrigation, and employing the plentiful sources of waste heat and CO_2. Integrating agriculture with urban planning is not a new concept, but deep consideration of energy use in the broader food system and the availability of relevant resources within cities (often as underexploited waste streams) can help realize substantial efficiency improvements in future urbanized food system.

Acknowledgments

This research was initiated through work completed during the National Science Foundation (NSF, grant number 1541838) funded workshop held October 5–6 at the University of Michigan entitled "Scaling' Up Urban Agriculture to Mitigate Food-Energy-Water-Impacts'. XF and LR acknowledge support from the NSF Sustainability Research Networks grant 1444745, and REFRESCH (Global Challenges for the Third Century program, Office of the Provost, University of Michigan). The authors thank Glen Daigger, Tim Dixon, Nancy Love, Josh Newell and Martin Sexton for comments on various iterations of this manuscript.

ORCID iDs

Eugene Mohareb https://orcid.org/0000-0003-0344-2253
Martin Heller https://orcid.org/0000-0001-9204-6222
Paige Novak https://orcid.org/0000-0001-9054-0278
Benjamin Goldstein https://orcid.org/0000-0003-0055-1323
Xavier Fonoll https://orcid.org/0000-0003-3304-2437
Lutgarde Raskin https://orcid.org/0000-0002-9625-4034

References

Ackerman K 2012 The potential for urban agriculture in New York City: growing capacity, food security and green infrastructure *Report* (New York: Columbia University Urban Design Lab) (http://urbandesignlab.columbia.edu/files/2015/04/4_urban_agriculture_nyc.pdf)

Ackerman K, Conard M, Culligan P, Plunz R, Sutto M P and Whittinghill L 2014 Sustainable food systems for future cities: the potential of urban agriculture *Econ. Soc. Rev.* **45** 189–206

Albaho M, Thomas B and Christopher A 2008 Evaluation of hydroponic techniques on growth and productivity of greenhouse grown bell pepper and strawberry *Int. J. Veg. Sci.* **14** 23–40

Alexander C, Ishikawa S, Silverstein M, Jacobson M, King I F and Angel S 1977 *A Pattern Language* (New York: Oxford University Press)

Almeida J, Achten W M J, Verbist B, Heuts R F, Schrevens E and Muys B 2014 Carbon and water footprints and energy use of greenhouse tomato production in Northern Italy *J. Ind. Ecol.* **18** 898–908

Althaus H-J 2003 *Process—steel production, converter, chromium steel* 18/8 RER (EcoInvent Database Version 3.3)

Altieri M A, Companioni N, Cañizares K, Murphy C, Rosset P, Bourque M and Nicholls C I 1999 The greening of the 'barrios': urban agriculture for food security in Cuba *Agric. Human Values* **16** 131–40

Amoah P, Drechsel P, Abaidoo R C and Henseler M 2007 Irrigated urban vegetable production in Ghana: microbiological contamination in farms and markets and associated consumer risk groups *J. Water Health* **5** 455–66

Andrews T, Williams V and Dalmeny K 2017 Sustainable Food Cities (http://sustainablefoodcities.org/)

Anonymous 2011 Aeroponics *American Heritage Dictionary of the English Language* 5th edn (http://search.credoreference.com/content/entry/hmdictenglang/aeroponics/0)

Anonymous 2017 Britannica Academic (www.academic.eb.com)

Arup and C40 Cities 2014 *Climate Action in Megacities—C40 Cities Baseline and Opportunities* vol 2 (http://issuu.com/c40cities/docs/c40_climate_action_in_megacities)

Audsley E *et al* 1997 *Harmonization of Environmental Life Cycle Assessment for Agriculture* AIR3-CT94-2028 *Community Research and Technological Development Programme in the field of Agriculture and Agro-Industry, Including Fisheries AIR 3* (Brussels, Belgium)

Barbosa G, Gadelha F, Kublik N, Proctor A, Reichelm L, Weissinger E, Wohlleb G and Halden R 2015 Comparison of land, water, and energy requirements of lettuce grown using hydroponic vs. conventional agricultural methods *Int. J. Environ. Res. Public Health* **12** 6879–91

Barthel S and Isendahl C 2013 Urban gardens, agriculture, and water management: sources of resilience for long-term food security in cities *Ecol. Econ.* **86** 224–34

Bass B and Baskaran B 2001 Evaluating rooftop and vertical gardens as an adaptation strategy for urban areas *National Research Council of Canada Report* 46737 (www.nps.gov/tps/sustainability/greendocs/bass.pdf)
Basset-Mens C, McLaren S J and Ledgard S 2007 Exploring a comparative advantage for New Zealand cheese in terms of environmental performance *LCA Foods Conference: 5th Int. Conf. (Gothenburg, Sweden, 26 April)*
Bernstad A and la Cour Jansen J 2011 A life cycle approach to the management of household food waste—a Swedish full-scale case study *Waste Manage.* **31** 1879–96
Binkley D, Harsh S, Wolf C A, Safferman S and Kirk D 2013 Electricity purchase agreements and distributed energy policies for anaerobic digesters *Energy Policy* **53** 341–52
Bøckman O C, Kaarstad O, Lie O H and Richards I 1990 *Agriculture and Fertilizers* (Oslo: Norsk Hydra)
Boulard T, Raeppel C, Brun R, Lecompte F, Hayer F, Carmassi G and Gaillard G 2011 Environmental impact of greenhouse tomato production in France *Agron. Sustain. Dev.* **31** 757–77
Brodt S, Kramer K J, Kendall A and Feenstra G 2013 Comparing environmental impacts of regional and national-scale food supply chains: a case study of processed tomatoes *Food Policy* **42** 106–14
Butler W H 2011 Welcoming animals back to the city: navigating public health tensions of urban livestock to achieve healthy and resilient communities *J. Agric. Food Syst. Community Dev.* **2** 193–215
Canning P, Charles A, Huang S, Polenske K R and Waters A 2010 Energy use in the US food system *Research Report #94* (Washington, DC: USDA ERS) (www.ers.usda.gov/publications/pub-details/?pubid=46377)
Carol M 2010 Ontario's water-energy nexus: will we find ourselves in hot water or …tap into opportunity? (Toronto: POLIS Research) *Report 10-01* (http://polisproject.org/files/pub_database/nexus-report_final.pdf)
Center for Sustainable Systems 2014 *US Wastewater Treatment* (Ann Arbor) (http://css.snre.umich.edu/css_doc/CSS04-14.pdf)
Cerón-Palma I, Sanyé-Mengual E, Oliver-Solà J, Montero J-I and Rieradevall J 2012 Barriers and opportunities regarding the implementation of rooftop ecogreenhouses (RTEG) in Mediterranean cities of Europe *J. Urban Technol.* **19** 1–17
Chen S 2012 Civic agriculture: towards a local food web for sustainable urban development *APCBEE Procedia* **1** 169–76
Clift R and Druckman A 2016 Industrial ecology's first decade *Taking Stock of Industrial Ecology* ed R Clift and A Druckman (London: Springer) ch 1 p 373
Coley D, Howard M and Winter M 2009 Local food, food miles and carbon emissions: a comparison of farm shop and mass distribution approaches *Food Policy* **34** 150–5
Curtis T P 2010 Low-energy wastewater treatment: strategies and technologies *Environmental Microbiology* 2nd edn ed R Mitchell and J D Gu (Hoboken, NJ: Wiley-Blackwell)
Delor M 2011 Current state of building-integrated agriculture, its energy benefits and comparison with green roofs–summary *Report* (http://bit.ly/1ihZebG)
Despommier D 2013 Farming up the city: the rise of urban vertical farms *Trends Biotechnol.* **31** 388–9
Edwards-Jones G *et al* 2008 Testing the assertion that 'local food is best' the challenges of an evidence based approach *Trends Food Sci. Technol.* **19** 265–74
Electric Power Research Institute 2002 Water and sustainability *US Electricity Consumption for Water Supply and Treatment–the Next Half Century* vol 4 (Palo Alto, CA: EPRI) (www.circleofblue.org/wp-content/uploads/2010/08/EPRI-Volume-4.pdf)
Elsayed M A, Matthews R and Mortimer N D 2003 *Carbon and Energy Balances for a Range of Biofuels Options—Project Number* B/B6/00784/REP URN 03/836
Energy Information Administration 2010 Trends in US residential natural gas consumption (www.eia.gov/pub/oil_gas/natural_gas/feature_articles/2010/ngtrendsresidcon/ngtrendsresidcon.pdf)
Esengun K, Erdal G, Gündüz O and Erdal H 2007 An economic analysis and energy use in stake-tomato production in Tokat province of Turkey *Renew. Energy* **32** 1873–81
Foley J A *et al* 2011 Solutions for a cultivated planet *Nature* **478** 337–42
Fonoll X, Astals S, Dosta J and Mata-Alvarez J 2015 Anaerobic co-digestion of sewage sludge and fruit wastes: evaluation of the transitory states when the co-substrate is changed *Chem. Eng. J.* **262** 1268–74
Food and Agriculture Organization 2013 Climate-Smart Agriculture Sourcebook (www.fao.org/docrep/018/i3325e/i3325e00.htm)
Food and Agriculture Organization 2010 FAO Statistical Yearbook (www.fao.org/docrep/015/am081m/am081m00.htm)
Food and Agriculture Organization 2001 Stakeholders, systems and issues in urban livestock keeping *Livestock Keeping in Urban Areas* (www.fao.org/docrep/004/y0500e/y0500e00.htm#toc)
Forkes J 2007 Nitrogen balance for the urban food metabolism of Toronto, Canada *Resour. Conserv. Recycles.* **52** 74–94
Francis C, Hansen T, Fox A, Hesje P, Nelson H, Lawseth A and English A 2012 Farmland conversion to non-agricultural uses in the US and Canada: current impacts and concerns for the future *Int. J. Agron. Sust.* **10** 8–24
Garfí M, Gelman P, Comas J, Carrasco W and Ferrer I 2011 Agricultural reuse of the digestate from low-cost tubular digesters in rural Andean communities *Waste Manage.* **31** 2584–9
Garnett T 1997 Farming the city: the potential for urban agriculture *Ecologist* **26** 299–307
Garnett T 1999 Urban agriculture in London: rethinking our food economy *Report* (www.ruaf.org/sites/default/files/London_1.PDF)
Garnett T 2011 Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? *Food Policy* **36** S23–32
Gibbs D and Deutz P 2007 Reflections on implementing industrial ecology through eco-industrial park development *J. Clean. Prod.* **15** 1683–95
Goldstein B, Hauschild M, Fernandez J and Birkved M 2017 Contributions of local agriculture to urban sustainability in the Northeast United States *Environ. Sci. Technol.* **51** 7340–9
Goldstein B, Hauschild M, Fernández J and Birkved M 2016a Testing the environmental performance of urban agriculture as a food supply in northern climates *J. Clean. Prod.* **135** 984–94
Goldstein B, Hauschild M, Fernández J and Birkved M 2016b Urban versus conventional agriculture, taxonomy of resource profiles: a review *Agron. Sustain. Dev.* **36** 9
Gorgolewski M, Komisar J and Nasr J 2017 Carrot City Initiative (www.ryerson.ca/carrotcity/)
Grard B J-P *et al* 2015 Recycling urban waste as possible use for rooftop vegetable garden *Futur. Food J. Food Agric. Soc.* **3** 21–34
Grewal S S and Grewal P S 2012 Can cities become self-reliant in food? *Cities* **29** 1–11
Hamilton A J, Burry K, Mok H-F, Barker S F, Grove J R and Williamson V G 2013 Give peas a chance? Urban agriculture in developing countries. A review *Agron. Sustain. Dev.* **34** 45–73
Hamm M W 2015 Feeding cities—with indoor vertical farms? *Food Clim. Res. Netw.* (http://fcrn.org.uk/fcrn-blogs/michaelwhamm/feeding-cities-indoor-vertical-farms) (Accessed: 6 September 2017)
Hansen T L 2006 Life cycle modelling of environmental impacts of application of processed organic municipal solid waste on agricultural land (Easewaste) *Waste Manage. Res.* **24** 153–66
Hanssen O J, Vold M, Schakenda V, Tufte P A, Møller H, Olsen N V and Skaret J 2017 Environmental profile, packaging intensity and food waste generation for three types of dinner meals *J. Clean. Prod.* **142** 395–402

Hatirli S A, Ozkan B and Fert C 2006 Energy inputs and crop yield relationship in greenhouse tomato production *Renew. Energy* **31** 427–38
Heidari M D and Omid M 2011 Energy use patterns and econometric models of major greenhouse vegetable productions in Iran *Energy* **36** 220–5
Heller M C and Keoleian G A 2015 Greenhouse gas emission estimates of US dietary choices and food loss *J. Ind. Ecol.* **19** 291–401
Horst M, McClintock N and Hoey L 2017 The intersection of planning, urban agriculture, and food justice: a review of the literature *J. Am. Plan. Assoc.* **83** 277–95
Hospido A, Milà I Canals L, McLaren S, Truninger M, Edwards-Jones G and Clift R 2009 The role of seasonality in lettuce consumption: a case study of environmental and social aspects *Int. J. Life Cycle Assess.* **14** 381–91
Howe J and Wheeler P 1999 Urban food growing: the experience of two UK cities *Sustain. Dev.* **7** 13–24
IPCC 2006 *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (www.ipcc-nggip.iges.or.jp/public/2006gl/)
Jaffal I, Ouldboukhitine S-E and Belarbi R 2012 A comprehensive study of the impact of green roofs on building energy performance *Renew. Energy* **43** 157–64
Jansson Å 2013 Reaching for a sustainable, resilient urban future using the lens of ecosystem services *Ecol. Econ.* **86** 285–91
Khan S, Cao Q, Zheng Y M, Huang Y Z and Zhu Y G 2008 Health risks of heavy metals in contaminated soils and food crops irrigated with wastewater in Beijing, China *Environ. Pollut.* **152** 686–92
Kimball B A 1983 Carbon dioxide and agricultural yield: an assemblage and analysis of 430 prior observations *Agron. J.* **75** 779–88
Klein G, Krebs M, Hall V, O'Brien T and Blevins B B 2005 California's water–energy relationship *California Energy Commission Report* CEC-700-2005-011-SF (www.energy.ca.gov/2005publications/CEC-700-2005-011/CEC-700-2005-011-SF.PDF)
Krannich J M 2006 A modern disaster: agricultural land, urban growth, and the need for a federally organized comprehensive land use planning model *Cornell J. Law Public Policy* **16** 56–99
Kulak M, Graves A and Chatterton J 2013 Reducing greenhouse gas emissions with urban agriculture: a life cycle assessment perspective *Landsc. Urban Plan.* **111** 68–78
Kuswardhani N, Soni P and Shivakoti G P 2013 Comparative energy input-output and financial analyses of greenhouse and open field vegetables production in West Java, Indonesia *Energy* **53** 83–92
Leach G 1975 Energy and food production *Food Policy* **1** 62–73
Levis J W and Barlaz M a 2011 What is the most environmentally beneficial way to treat commercial food waste? *Environ. Sci. Technol.* **45** 7438–44
Loder A 2014 There's a meadow outside my workplace: a phenomenological exploration of aesthetics and green roofs in Chicago and Toronto *Landsc. Urban Plan.* **126** 94–106
Mamais D, Noutsopoulos C, Dimopoulou A, Stasinakis A and Lekkas T D 2015 Wastewater treatment process impact on energy savings and greenhouse gas emissions *Water Sci. Technol.* **71** 303–8
Martellozzo F, Landry J-S, Plouffe D, Seufert V, Rowhani P and Ramankutty N 2014 Urban agriculture: a global analysis of the space constraint to meet urban vegetable demand *Environ. Res. Lett.* **9** 064025
Martínez-Blanco J, Muñoz P, Antón A and Rieradevall J 2011 Assessment of tomato Mediterranean production in open-field and standard multi-tunnel greenhouse, with compost or mineral fertilizers, from an agricultural and environmental standpoint *J. Clean. Prod.* **19** 985–97
McClintock N, Cooper J and Khandeshi S 2013 Assessing the potential contribution of vacant land to urban vegetable production and consumption in Oakland, California *Landsc. Urban Plan.* **111** 46–58
McCurry D, Bear S, Bae J, Sedlak D, McCarty P and Mitch W 2014 Superior removal of disinfection byproduct precursors and pharmaceuticals from wastewater in a staged anaerobic fluidized membrane bioreactor compared to activated sludge *Environ. Sci. Technol. Lett* **1** 459–64
Metson G, Aggarwal R and Childers D L 2012 Efficiency through proximity: changes in phosphorus cycling at the urban-agricultural interface of a rapidly urbanizing desert region *J. Ind. Ecol.* **16** 914–27
Milan C 2015 Milan Urban Food Policy Pact (www.foodpolicymilano.org/en/urban-food-policy-pact-2/)
Mohareb E A, MacLean H L and Kennedy C A 2011 Greenhouse gas emissions from waste management—assessment of quantification methods *J. Air Waste Manage. Assoc.* **61** 480–93
Mok H-F F, Williamson V G, Grove J R, Burry K, Barker S F and Hamilton A J 2013 Strawberry fields forever? Urban agriculture in developed countries: a review *Agron. Sustain. Dev.* **34** 21–43
Møller J, Boldrin A and Christensen T H 2009 Anaerobic digestion and digestate use: accounting of greenhouse gases and global warming contribution *Waste Manage. Res.* **27** 813–24
Monlau F, Francavilla M, Sambusiti C, Antoniou N, Solhy A, Libutti A, Zabaniotou A, Barakat A and Monteleone M 2016 Toward a functional integration of anaerobic digestion and pyrolysis for a sustainable resource management. Comparison between solid-digestate and its derived pyrochar as soil amendment *Appl. Energy* **169** 652–62
Mortensen L M 1987 Review: CO_2 enrichment in greenhouses. Crop responses *Sci. Hortic.* **33** 1–25
Mortimer N D, Cormack P, Elsayed M A and Horne R E 2003 Evaluation of the comparative energy, global warming and socio-economic costs and benefits of biodiesel (http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=10701)
Nicholson C F, He X, Gómez M I, Gao H O and Hill E 2015 Environmental and economic impacts of localizing food systems: the case of dairy supply chains in the Northeastern United States *Environ. Sci. Technol.* **49** 12005–14
Oke T R 1973 City size and the urban heat island *Atmos. Environ.* **7** 769–79
Orsini F, Gasperi D, Marchetti L, Piovene C, Draghetti S, Ramazzotti S, Bazzocchi G and Gianquinto G 2014 Exploring the production capacity of rooftop gardens (RTGs) in urban agriculture: the potential impact on food and nutrition security, biodiversity and other ecosystem services in the city of Bologna *Food Secur.* **6** 781–92
Orsini F, Kahane R, Nono-Womdim R and Gianquinto G 2013 Urban agriculture in the developing world: a review *Agron. Sustain. Dev.* **33** 695–720
Pagano M A and Bowman A O 2000 Vacant land in cities: an urban resource *Brookings Institute Report* (www.brookings.edu/wp-content/uploads/2016/06/paganofinal.pdf)
Pataki D E *et al* 2011 Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions *Front. Ecol. Environ.* **9** 27–36
Payen S, Basset-Mens C and Perret S 2015 LCA of local and imported tomato: an energy and water trade-off *J. Clean. Prod.* **87** 139–48
Pelletier N, Audsley E, Brodt S, Garnett T, Henriksson P, Kendall A, Kramer K J, Murphy D, Nemecek T and Troell M 2011 Energy intensity of agriculture and food systems *Annu. Rev. Environ. Resour.* **36** 223–46
Pimentel D, Williamson S, Alexander C E, Gonzalez-Pagan O, Kontak C and Mulkey S E 2008 Reducing energy inputs in the US food system *Hum. Ecol.* **36** 459–71
Pirog R, Van Pelt T, Enshayan K and Cook E 2001 Food, fuel, and freeways: an Iowa perspective on how far food travels, fuel usage, and greenhouse gas emissions *Report* (http://lib.dr.iastate.edu/cgi/viewcontent.cgi?article=1002&context=leopold_pubspapers)
Plawecki R, Pirog R, Montri A and Hamm M W 2014 Comparative carbon footprint assessment of winter lettuce production in two climatic zones for Midwestern market *Renew. Agric. Food Syst.* **29** 310–8

Racoviceanu A I, Karney B W, Kennedy C A and Colombo A F 2007 Life-cycle energy use and greenhouse gas emissions inventory for water treatment systems *J. Infrastruct. Syst.* **13** 261–70

Ramaswami A, Boyer D, Nagpure A S, Fang A, Bogra S, Bakshi B, Cohen E and Rao-Ghorpade A 2017 An urban systems framework to assess the trans-boundary food–energy–water nexus: implementation in Delhi, India *Environ. Res. Lett.* **12** 025008

Riva C, Schievano A, D'Imporzano G and Adani F 2014 Production costs and operative margins in electric energy generation from biogas. Full-scale case studies in Italy *Waste Manage.* **34** 1429–35

Romero-Guiza M S, Peces M, Astals S, Benavent J, Valls J and Mata-Alvarez J 2014 Implementation of a prototypal optical sorter as core of the new pre-treatment configuration of a mechanical-biological treatment plant treating OFMSW through anaerobic digestion *Appl. Energy* **135** 63–70

Saiz S, Kennedy C, Bass B and Pressnail K 2006 Comparative life cycle assessment of standard and green roofs *Environ. Sci. Technol.* **40** 4312–6

Sanscartier D, Maclean H L and Saville B 2012 Electricity production from anaerobic digestion of household organic waste in Ontario: techno-economic and GHG emission analyses *Environ. Sci. Technol.* **46** 1233–42

Sanyé-Mengual E, Oliver-Solà J, Montero J I and Rieradevall J 2015 An environmental and economic life cycle assessment of rooftop greenhouse (RTG) implementation in Barcelona, Spain. Assessing new forms of urban agriculture from the greenhouse structure to the final product level *Int. J. Life Cycle Assess.* **20** 350–66

Schafft K A, Jensen E B and Clare Hinrichs C 2009 Food deserts and overweight schoolchildren: evidence from Pennsylvania *Rural Sociol.* **74** 153–77

Shiina T, Hosokawa D, Roy P, Orikasa T, Nakamura N and Thammawong M 2011 Life cycle inventory analysis of leafy vegetables grown in two types of plant factories *Acta Hortic.* **919** 115–22

Shrestha S, Fonoll X, Khanal S K and Raskin L 2017 Biological strategies for enhanced hydrolysis of lignocellulosic biomass during anaerobic digestion: current status and future perspectives *Bioresour. Technol.* **245** 1245–57

Smil V 2004 Improving efficiency and reducing waste in our food system *Environ. Sci.* **1** 17–26

Smit J and Nasr J 1992 Urban agriculture for sustainable cities: using wastes and idle land and water bodies as resources *Environ. Urban.* **4** 141–52

Smith A, Brown K, Ogilvie S, Rushton K and Bates J 2001 Waste management options and climate change: final report to the European Commission

Smith A, Stadler L, Cao L, Love N, Raskin L and Skerlos S 2014 Navigating wastewater energy recovery strategies: a life cycle comparison of anaerobic membrane bioreactor and conventional treatment systems with anaerobic digestion *Environ. Sci. Technol.* **48** 5972–81

Smith A, Stadler L, Love N, Skerlos S and Raskin L 2012 Perspectives on anaerobic membrane bioreactor treatment of domestic wastewater: a critical review *Bioresour. Technol.* **122** 149–59

Specht K, Siebert R, Hartmann I, Freisinger U B, Sawicka M, Werner A, Thomaier S, Henckel D, Walk H and Dierich A 2014 Urban agriculture of the future: an overview of sustainability aspects of food production in and on buildings *Agric. Human Values* **31** 33–51

Tamburini E, Pedrini P, Marchetti M, Fano E and Castaldelli G 2015 Life cycle based evaluation of environmental and economic impacts of agricultural productions in the mediterranean area *Sustainability* **7** 2915–35

Taylor J R and Lovell S T 2012 Mapping public and private spaces of urban agriculture in Chicago through the analysis of high-resolution aerial images in Google Earth *Landsc. Urban Plan.* **108** 57–70

Thebo A L, Drechsel P and Lambin E F 2014 Global assessment of urban and peri-urban agriculture: irrigated and rainfed croplands *Environ. Res. Lett.* **9** 114002

US Global Change Research Program 2014 *Climate Change Impacts in the United States: The Third National Climate Assessment* (nca2014.globalchange.gov%5CnThis)

Uçkun Kiran E, Trzcinski A P, Ng W J and Liu Y 2014 Bioconversion of food waste to energy: a review *Fuel* **134** 389–99

US Census Bureau 2015 2010 Census Urban and Rural Classification and Urban Area Criteria (www.census.gov/geo/reference/ua/urban-rural-2010.html)

US Department of Agriculture 2014 US census of agriculture *National Level Data* vol 1 (www.agcensus.usda.gov/Publications/2012/Full_Report/Volume_1,_Chapter_1_US/)

US Environmental Protection Agency 2008 Ensuring a sustainable future: an energy management guidebook for wastewater and water utilities *Report* (https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P1003Y1G.TXT)

US Environmental Protection Agency 2010 Evaluating the environmental impacts of packaging fresh tomatoes using life-cycle thinking and assessment: a sustainable materials management demonstration project *Report* (www.epa.gov/wastes/conserve/tools/stewardship/docs/tomato-packaging-assessment.pdf)

US EPA 2015 eGRID tables—2012 (www.epa.gov/energy/egrid)

US EPA 2017 Inventory of US greenhouse gas emissions and sinks: 1990–2015 *Report* (Washington, DC) (www.epa.gov/sites/production/files/2017-02/documents/2017_complete_report.pdf)

USDA ERS 2010 Energy use in the US food system *Report* (www.ers.usda.gov/media/136418/err94_1_.pdf)

USDA ERS 2014 Farm labor (www.ers.usda.gov/topics/farm-economy/farm-labor/)

USDA ERS 2013 Food Availability *Data Syst.* (www.ers.usda.gov/data-products/food-availability-(per-capita)-data-system.aspx)

Vitiello D and Brinkley C 2013 The hidden history of food system planning *J. Plan. Hist.* **13** 91–112

Wakefield S, Yeudall F, Taron C, Reynolds J and Skinner A 2007 Growing urban health: community gardening in South-East Toronto *Health Promot. Int.* **22** 92–101

Ward J D 2015 Can urban agriculture usefully improve food resilience? Insights from a linear programming approach *J. Environ. Stud. Sci.* **5** 699–711

Webb J, Williams A G, Hope E, Evans D and Moorhouse E 2013 Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK? *Int. J. Life Cycle Assess.* **18** 1325–43

Weber C L and Matthews H S 2008 Food-miles and the relative climate impacts of food choices in the United States *Environ. Sci. Technol.* **42** 3508–13

West T O and Marland G 2002 A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States *Agric. Ecosyst. Environ.* **91** 217–32

Williams H and Wikstrom F 2011 Environmental impact of packaging and food losses in a life cycle perspective: a comparative analysis of five food items *J. Clean. Prod.* **19** 43–8

World Health Organization 2006 Guidelines for the safe use of wastewater, excreta and greywater *Report* vol 1 (Geneva: WHO) (http://whqlibdoc.who.int/publications/2006/9241546832_eng.pdf)

Zhang S, Bi X T and Clift R 2013 A life cycle assessment of integrated dairy farm-greenhouse systems in British Columbia *Bioresour. Technol.* **150** 496–505