Best practices for consistent and reliable life cycle assessments of urban agriculture

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How to do life cycle assessments of urban agriculture

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Highlights

- Life cycle assessment is inconsistently applied to urban agriculture
- Identified key challenges of doing life cycle assessment of this unique activity
- Made practical recommendations for how to address these challenges
- Outlined research directions and scientific questions for this maturing topic
- Following recommendations will strengthen and clarify body of research

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Abstract

There is increasing interest in evaluating the environmental performance of urban agriculture (UA), especially using life cycle assessment (LCA). However, LCA has been applied to UA inconsistently, making it difficult to confidently compare or draw conclusions from existing studies. Here, we outline the key challenges of applying LCA to UA and recommend concrete steps to help bring consistency and comprehensiveness to the topic. First, we clarify the research questions that can be addressed with LCA. We then provide practical recommendations for performing LCAs of UA, considering several of its unique aspects that require special attention by practitioners. These include crop diversity, data availability, modeling compost, soil carbon sequestration, producing growing media, distribution of crops, and variability and uncertainty. Next, we propose future research areas that will benefit LCA generally and its application to UA, such as framing UA as urban green infrastructure, evaluations at the city-scale, accounting for ecosystem services, and including social dimensions of UA. By following these recommendations, future LCAs of UA can be more consistent, comparable, and holistic, and will help build knowledge and inform policy making and practices around UA.

Keywords: LCA guidelines, farms, gardens, social LCA, ecosystem services, compost

1 Introduction

Urban agriculture (UA) is a multifunctional activity with many assumed and demonstrated benefits for cities and their inhabitants. These social, economic, and environmental benefits position UA as a powerful tool to improve urban environments, contribute to more sustainable of urban food systems, and enhance wellbeing of urban dwellers (Azunre et al., 2019; Gómez-Villarino et al., 2021). Food grown in cities can have lower environmental burdens than food from conventional farms for a variety of indicators, including site-specific pollution to diffuse greenhouse gas (GHG) emissions (Nicholls et al., 2020; O'Sullivan et al., 2019). While pollution at farms can be measured on site, environmental footprinting methods, such as life cycle assessment (LCA), are needed to capture impacts across the food value chain. Although the LCA method is standardized, findings from available LCAs of UA are highly variable because of inconsistencies in how the method has been employed (Dorr et al., 2021a). We lack reliable answers to important questions surrounding the environmental performance of UA. What types of UA have lower impacts than others? What are the main sources of impacts in UA? Can UA help reduce the environmental impacts of the food system? Researchers require guidance to more consistently make decisions regarding system modeling, system boundaries, and reporting so that LCAs of UA can help answer these and other questions.

General LCA frameworks and guides have been proposed to improve the rigor and comparability of LCAs, and include the International Organization for Standardization (ISO) framework (ISO, 2006a, 2006b), the ILCD handbook (European Commission, 2010a), and the Product Environmental Footprint Category Rules Guidance (European Commission, 2017). For LCA of specific sectors, methodological guidelines have highlighted unique aspects that require special attention. Failure to account for these aspects can skew results and hamper decision making. For instance, the inclusion of direct and indirect land use change in biofuel production fundamentally altered the carbon calculus of this technology, and caused a reappraisal of government policies to support first-generation biofuels (Searchinger et al., 2008).

To avoid similar mistakes in other fields, researchers have produced LCA guidelines for diverse industries and technologies ranging from waste management (Laurent et al., 2014) to bioplastics (Bishop et al., 2021). In the area of food, best practices exist for LCAs of crop production (Adewale et al., 2018), organic agriculture (van der Werf et al., 2020), fruit orchards (Cerutti et al., 2014), vegetables (Perrin et al., 2014), climate-smart agriculture (Acosta-Alba et al., 2019) and agricultural use of microbial inoculants (Kløverpris et al., 2020). Other work has evaluated the combination of agricultural LCAs with circular economy (Stillitano et al., 2021) or ecosystem service assessments (Tang et al., 2018). Not to mention the large body of work reviewing the methodological choices, challenges, and best practices of agricultural LCAs in general (Audsley et al., 1997; Brentrup et al., 2004; Caffrey and Veal, 2013; Cucurachi et al., 2019; Dijkman et al., 2018; Mclaren, 2010; Nemecek and Gaillard, 2010; Notarnicola et al., 2017, 2012). Such methodological reflections and recommendations have not yet been done for UA.

This study intends to fill this gap by providing a guideline for how to assess UA using LCA. It is applicable to all forms of UA in its most general definition of “food production in and around cities” (Mougeot, 2000). It builds on observations from a previous literature review and meta-analysis of the environmental impacts of UA (Dorr et al., 2021a) to provide practical recommendations when applying LCA to UA, and takes a more comprehensive approach to both UA and LCA. This guideline was also tested and iteratively refined during a recent LCA of a diverse set of urban farms in France and the United States (Dorr et al., 2022b).

This paper begins by reflecting on the goals and expectations of LCAs of UA, followed by practical recommendations to make LCAs of UA more consistent, and research directions to improve LCAs of UA. In doing so, this paper identifies the challenges of including certain aspects of UA in LCA, reviews how these aspects are currently treated in LCAs of UA, and recommendations for how to best treat them going forward. This guideline is intended to complement existing frameworks for agricultural LCAs, and some issues relevant to both

LCA of conventional agriculture and UA were included here. Our hope is that by outlining clear rules for dealing with the unique challenges of applying LCA to UA, future work can be done in an a consistent, transparent, and comprehensive manner. Such consistency is needed to determine under what conditions and in what forms UA can meaningfully contribute to urban sustainability.

2 Why do LCAs of urban agriculture?

Since there are diverse framings of UA, it is useful to clarify why we should study it with LCA, by defining both the goals and the larger questions they aim to answer. Reflecting on these questions is especially timely as UA LCAs evolve from an early stage with relatively simple goals of assessing impacts of a farm or garden, to a more mature stage assessing more complex topics. The goal of an LCA dictates how the assessment is set up. All decisions regarding system boundaries, functional unit (FU), and interpretations should be consistent with the defined goal(s) of the study, which should reflect the pursuit of an overarching question(s). Table 1 highlights some key, largely unanswered questions around UA that LCA can address. Goals of existing UA LCAs include evaluating the environmental impacts of urban food production at the farm-scale, identifying ways to reduce these impacts, comparing UA to rural agriculture or to other urban land uses, comparing types of UA, and evaluating the consequences of developing UA (such as reduced lawn management, or municipal treatment of organic waste) ((Dorr et al., 2021a). A more detailed review of UA LCAs that addressed each question, with goal, scope, and FU recommendations, is in Appendix A.

Table 1 The goal of a life cycle assessment should answer a larger question. Some relevant questions for life cycle assessment of urban agriculture are presented here, along with a description/justification for each question and possible functional units (FU). Some questions are already prevalent in the literature, and some are our original suggestions and have not been addressed before.

Question	Description	FU
Is UA an environmentally positive way to feed the city, relative to the status quo of conventional food systems?	In light of new urban food planning strategies, and initiatives to reduce impacts of public food procurement, we should investigate if UA is a useful strategy.	Single crop, mixed crops, cost/revenue, individual diet, city-wide food flows
Is UA an environmentally positive type of green	Green infrastructure is promoted in cities, and many types are possible.	Land area, cost/revenue

infrastructure to implement in a city?	City leaders must decide which types to implement.	
How does UA affect the GHG emission or other environmental impacts of a city?	Cities have pledged to reduce GHG emissions, which UA may address through land use, replacing other food sources, changing consumers' behaviors, or altering organic waste treatment.	Urban metabolism, land area, operation of other sectors (i.e. waste treatment)
What are potential trade-offs of socially motivated UA projects?	Many UA projects do not claim to have environmental motivations or particularly low impacts, but they are promoted based on other merits (often social). Are there important tradeoffs between the social and environmental dimensions? Can we justify an environmentally harmful activity if it delivers social benefits?	Single crop, mixed crops, land area, cost/revenue, total operations of urban farm, social functions (i.e. hours of education, number of participants)
Which type of UA should be developed/promoted for a given motivation (indoor or outdoor, hydroponics or soil-based, commercial or non-profit, professional or volunteer-based…)?	Developers, city leaders, and stakeholders may have land that they want to dedicate to UA. With the vast diversity of types of UA, they may need support deciding which type to develop, according to environmental and other dimensions.	Single crop, mixed crops, land area, cost/revenue, total operations of urban farm, social functions (i.e. hours of education, number of participants)
How can UA be designed or managed to minimize environmental impacts?	In many cases, UA will be practiced regardless of the above questions. Then, we should inform practitioners of the best practices to minimize their impact.	Any

3 Challenges and practical recommendations for UA LCAs

Below, we describe unique aspects of UA that present methodological challenges for LCAs, and our recommendations for addressing them. Each section includes an explanation of the challenge, examples of how it has been treated in previous urban or rural agriculture LCAs, and recommendations for future work. Section 3.3, on compost, includes additional subsections because there are numerous challenges, and to the best of our knowledge its inclusion in agricultural LCAs has not been reviewed before. A summary of key recommendations is provided in Table 2, which draws from both the practical recommendations here and the research directions presented in section 4.

3.1 Crop diversity

Challenge:

Mass-based FUs are most common in crop production LCAs (Notarnicola et al., 2017). For monoculture farms, there are no allocation issues: all inputs and impacts are assigned to one crop. For farms growing multiple crops either with temporal diversity (crop rotation) or spatial diversity (polyculture/intercropping), allocation between crops is needed (Adewale et al., 2018). For polycultures, rural/professional farmers can often specify which inputs were used on various farm parcels, and fixed inputs can be allocated by mass, revenue or other measure (Caffrey and Veal, 2013). For crop rotations, allocation principals have been proposed (Brankatschk and Finkbeiner, 2015). Such allocation is difficult for UA, where crop diversity is often exceedingly high: urban farms may cultivate on average 20-30 crops per year, with extremes of 80-130 (Gregory et al., 2016; Kirkpatrick and Davison, 2018; Pourias et al., 2016). It is unreasonable to expect urban farmers to distinguish inputs for so many crops, so LCA practitioners often contend with the challenge of including many crops in one FU. This issue is not unique to UA—it is also relevant for diversified rural farms and community-supported agriculture (CSA) (Caffrey and Veal, 2013; Christensen et al., 2018)—but is more pronounced with UA.

Examples:

LCA stage	No.	Recommendation
Goal and scope	1	Be transparent, thorough, and critical when evaluating compost, substrate, and other organic inputs. They are especially important for UA, and are not usually the focus in agricultural LCAs.
	2	Use multiple FUs—at least land and product-based.
	3	Include post-farm transport of products—especially the (near) zero impacts of transport by bike or on foot.
	4	Account for seasonality, local context, and (where relevant) last mile transport for more precise comparisons to rural agriculture.
Life cycle inventory	5	Collect primary data from functioning urban farms, because UA may not operate as expected or as measured under ideal, controlled conditions.
Life cycle impact assessment	6	Use sensitivity analyses for important parameters with high uncertainty or variability to obtain a range or distribution of results. Such parameters may be related to: • Infrastructure lifetime • Substrate lifetime • Compost emission factors • Delivery logistics
	7	Present results with and without major avoided burdens and carbon sequestration benefits.
Interpretation	8	Provide more holistic descriptions of UA case studies and their urban contexts, because UA is diverse and vaguely defined. This includes the motivations, management/farming structures, or innovative status of a case study.
	9	Compare impacts with an area-based FU to other urban green infrastructure or urban land use options.
	10	Include social, economic, and ecosystem service-related measures, even if they are not life-cycle based.

Table 2 Ten key recommendations for performing UA LCAs are summarized according to their position along the 4-step LCA process.

Most UA LCAs with high crop diversity chose FUs covering total annual operations of a farm or impacts per unit area (Martinez et al., 2018; Pérez-Neira and Grollmus-Venegas, 2018; Sanyé-Mengual et al., 2018a). This avoids highly uncertain allocations, considers additional functions of agriculture, and facilitates cross-farm comparisons. However, results are difficult to extrapolate since they represent production of varied crops which are usually not functionally comparable, and sometimes the crops grown are not communicated. Another strategy uses published data or farmer estimates to estimate an life cycle inventory for each crop (Caputo et al., 2020; Kulak et al., 2013; Liang et al., 2019). This allows for crop-level analysis for polycultures, but accuracy is inevitably lost when equating UA to other systems. For instance, when these data come from rural agriculture, representativeness of UA is likely sacrificed. Other researchers have allocated between many crops based on mass, area, calorie content, nutritional index, or time of cultivation of each crop, to generate results per crop (Pennisi et al., 2019; Rufí-Salís et al., 2020a; Sanyé-Mengual et al., 2015b). Finally, some researchers used a simplified FU covering a basket of crops (i.e. 1 kilogram of mixed lettuce, tomato, and pepper) (Boneta et al., 2019; Hu et al., 2019). These results are difficult to use elsewhere since unique mixes of crops are not precisely comparable, and authors may not include which crops are included in the mix or in what proportions. LCAs of rural farms with many crops have also used a FU of kilogram of mixed crop (Christensen et al., 2018; Pepin, 2022), which complicates interpretation.

Recommendations:

The main options for dealing with multi-crop UA systems are to evaluate a basket of products (by mass or by converting to calories or nutritional indexes), allocate between products, or choose a FU that is not based on food production. It is impossible to universally recommend a FU for LCAs of such diverse systems aiming to answer different questions, and ultimately the choice of FU depends on the goal of the LCA, but we can recommend some best practices. When a FU other than single crop is used, a breakdown of how much of each crop was grown should be provided, to give some indication of what the food outputs of the system were. Ideally UA LCAs should aim for crop-specific inventories within urban farms, to allow for a FU of production of a single crop, but due to high crop biodiversity this may not be feasible. Finally, providing results across multiple FUs can illuminate tradeoffs and compensate for the opaque nature of mixed-product FUs such as mass of mixed crops.

3.2 Data availability

Challenge:

Data collection in LCA is often highly labor-intensive. For an agricultural LCA, data on farm inputs and outputs are needed. In conventional agriculture, primary data come from farmer interviews, receipts, or informed estimates/calculations (Christensen et al., 2018). Secondary data, such as the UC Davis Crop Budgets (Caffrey and Veal, 2013), can address data gaps or create entire inventories. Similar quality data are rare for UA because urban farmers usually keep limited records (Cleveland, 1997; Egerer et al., 2018; Whittinghill and Sarr, 2021). Inputs and food production in UA (especially informal UA) can be extremely variable and difficult to predict, casting doubt on the applicability of secondary data for UA (Dorr et al., 2022c). Collective and community-based UA may have many participants who harvest and use agricultural inputs, which further complicates record keeping. Self-reporting and participatory methods face issues of reliable, consistent data collection and participant fatigue (CoDyre et al., 2015).

Examples:

The available UA LCAs are based on both primary and secondary data. Data for UA LCAs come from many different sources, including directly measured data, operations records, farmer interviews and surveys, and secondary data from urban or rural agriculture (Dorr et al., 2021a). Data sources and data collection difficulties are largely discussed in research on UA practices in general, but not so much in UA LCAs (McDougall et al., 2019; Pollard et al., 2018).

Recommendations:

Due to the variability and lack of data regarding UA practices, collecting primary data from case studies should be prioritized. Past records of operation may be used, although it is unlikely that urban farmers have records of all necessary information for an LCA. A data collection campaign, with commitment from farmers, may be necessary. Researchers should discuss data needs with farmers early and often to identify the most feasible methods to collect data, create a data collection plan, and regularly follow up to ensure reliability. This is a crucial step because unclear or overly burdensome data collection efforts may be abandoned or unusable. Researchers should consider the types of data that may already be collected at urban farms (i.e. level of detail, time frame, units), and adapt the data collection plan accordingly. Surveys, growing logs, and harvest notebooks should be co-designed with farmers to track harvest and inputs (Nicholls et al., 2020). Water use should be measured using water meters or calculated using the number of buckets or watering cans used and their volume (Pollard et al., 2018). Researchers should periodically check for leaks in irrigation systems, which may be substantial (Dorr et al., 2022c). Soil amendments, such as compost and fertilizers, should be tracked through the amount applied, or the amount purchased/delivered (although this may require temporal allocation to growing season). The detailed description of our data collection methods with UA case studies in the appendix of (Dorr et al., 2022b) provides concrete examples of how to collect data across diverse systems.

3.3 Compost

Compost is the main input to many urban farms (see detailed review in the Appendix B) (Cofie et al., 2006; Dobson et al., 2021; Edmondson et al., 2014). A proposed environmental advantage of UA is its potential to grow food and reduce landfill burdens by applying compost from urban organic waste (Mohareb et al., 2017; Specht et al., 2014). Compost is thus central to UA despite infrequent and inconsistent quantification in UA LCAs (Dorr et al., 2021a). Even for rural agriculture LCAs, compost is often omitted, or its inclusion is inconsistent and unclear (Bartzas et al., 2015). Surprisingly, compost is not explicitly mentioned in reviews of LCAs of organic agriculture, where it is expected to be extensively used (Meier et al., 2015; van der Werf et al., 2020). LCAs focusing on compost use in agriculture found that the GHGs emitted from microbial decomposition (CH_4 and N_2O) are a major contributor to climate change impacts, and avoided burdens (i.e. subtracting emissions from avoided processes, such as avoided incineration of organic waste) and allocation have large effects on the results for rural agriculture (Bartzas et al., 2015; Christensen et al., 2018; Martínez-Blanco et al., 2009) and for UA (Dorr et al., 2022b, 2017; Liang et al., 2019; Martin et al., 2019). Therefore, compost is given extra attention for this section.

3.3.1 Off-farm compost system modeling

Challenge:

Off-farm compost refers to compost purchased from municipal or industrial composting facilities, as opposed to on-farm compost, described below. In the authors' experiences, the majority of compost used in UA is purchased, because urban farms do not have the capacity to make sufficient quantities of compost on-farm. Off-farm compost used in UA is a recycled input, similar to using recycled plastic materials or recycled paper. Accounting for recycled inputs is a distinct allocation issue with a complicated and contested history in LCA (Frischknecht, 2010; Huppes and Curran, 2012; Toniolo et al., 2017; Weidema, 2000).

Examples:

A common practice to address recycling in LCA is the "simple cut off" method (Ekvall and Tillman, 1997). Here, the recycled product is cut off from the system that generated the waste, and enters the following system boundary when the waste material is transported to a recycling plant (Frischknecht, 2010). No impacts from the virgin material (for compost, this would be food or biomass production) are given to the system using the recycled product. Impacts of the recycling process and transport to the user are given to the system using the recycled material. This method can be refined by allocating some impacts from the recycling process to the upstream waste generator, considering the waste as a co-product that goes on to make a new good (Ekvall and Tillman, 1997). The ILCD Handbook (section 14.4.1.3) recommends this allocation method, considering that a valuable co-product is generated from the waste treatment process, and it is "inappropriate to attribute all preceding waste treatment processes to the eventually produced secondary good" (European Commission, 2010b). After allocating processes based on physical causality, an economic allocation is the preferred method to distribute impacts between the first system (i.e. that produced the waste) and the second system (i.e., the one that uses the compost) (European Commission, 2010b; Guinée et al., 2004). For compost, this has been done using the relative revenue at a composting plant between waste dumping fees and compost purchases (Christensen et al., 2018; Pepin, 2022).

For UA LCAs where off-farm compost was used, system modeling decisions have been mixed. In most cases, off-farm compost was included using the simple cut-off approach, giving all impacts to the compost product, with no avoided burdens (Goldstein et al., 2016; Ledesma et al., 2020; Liang et al., 2019; Martin et al., 2019; Rothwell et al., 2016).

Recommendations:

We recommend treating off-farm compost as a recycled input, using the refined cut off method to give compost no impacts from the virgin material production and some impacts from the composting process (Figure 1 and 2). Impacts from composting should be allocated between organic waste treatment (assigned to the waste generator) and compost production (assigned to the compost user). Avoided burdens of fertilizer production should be credited to the waste generator, and not the farm using compost, because the waste generator made the decision that led to creation of the product displacing fertilizer (Schrijvers et al., 2016).

3.3.2 On-farm compost system modeling

Challenge:

On-farm compost refers to the composting operations in a farm, mainly for composting inedible plant biomass. There are several possible scenarios for on-farm compost and consequently several modeling options (Figure 1 and 2). On-farm compost may be:

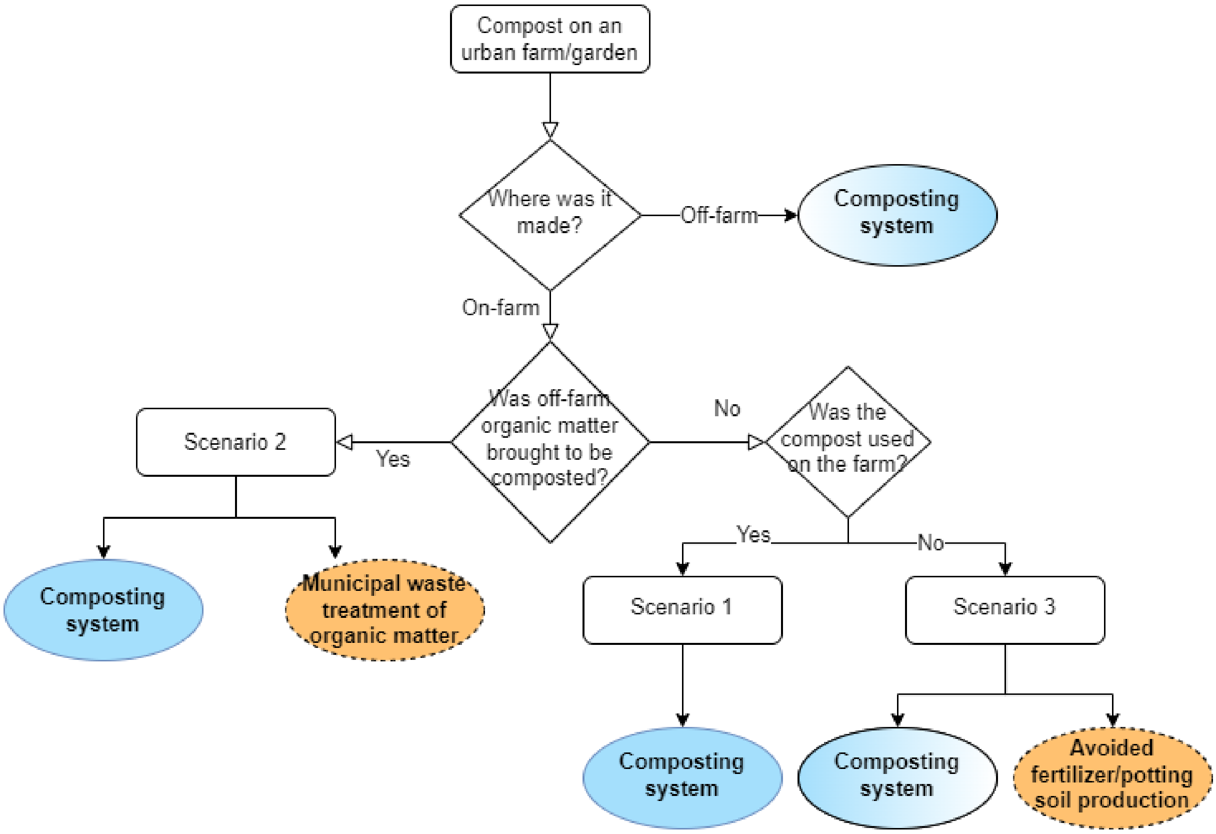


Figure 1 A decision tree clarifies the different scenarios of composting for an urban farm and how to account for composting impacts. Blue circles represent impacts from composting emissions, and orange circles with dotted outlines represent substituted processes that can be subtracted from the farm's impacts, thanks to composting. Blue circles with gradients represent the fact that not all impacts from composting in that scenario will go to the farm: they should be allocated between the organic waste producer and the compost user. The numbered scenarios are detailed in section 3.3.2.

Scenario 1) made using on-farm biomass and used on the farm,

Scenario 2) made using on-farm biomass plus other green waste brought to the farm, and used on the farm, or

Scenario 3) made using on-farm biomass and not used on the farm (i.e., for hydroponics systems that generate biomass waste but do not use compost).

These possible scenarios, and the relevant system modeling decisions for LCA, have not been explicitly examined before.

Examples and recommendations:

Scenario 1 is a type of "closed loop" recycling system where the waste is generated and the recycled product is used within the same system. Examples of this are in Boneta et al. (2019)

and Sanyé-Mengual et al. (2015b). System modeling is straightforward with all impacts from composting given to the farm, with no avoided burdens or allocation (ISO, 2006b).

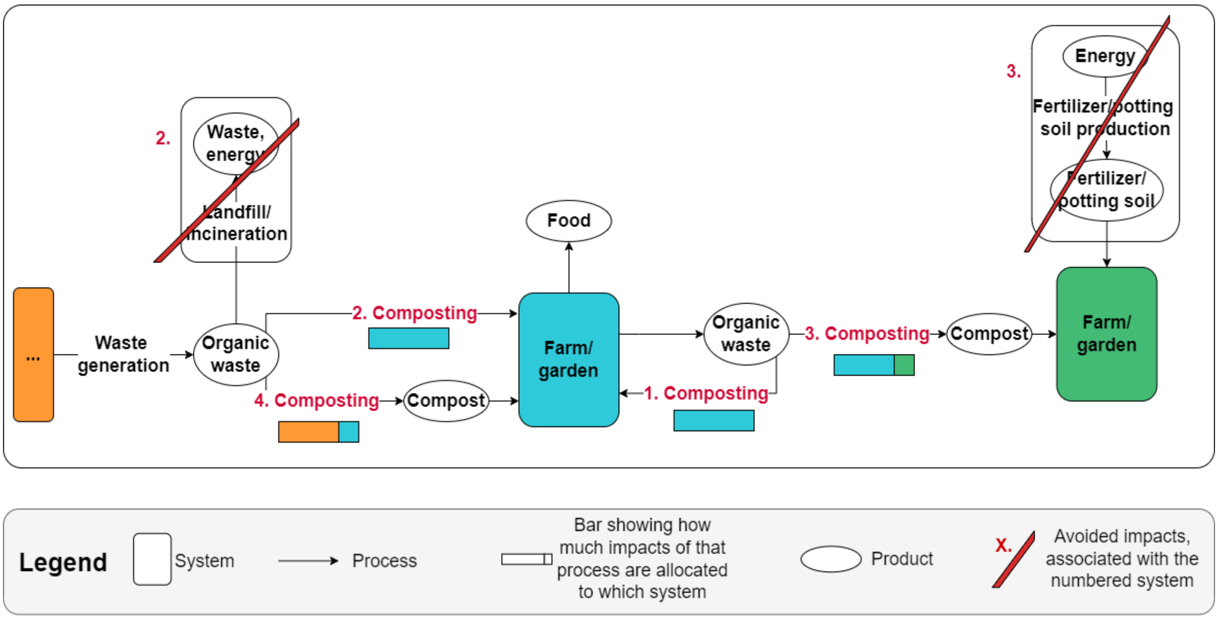


Figure 2 A process diagram shows the different composting scenarios as described in the text, from the perspective of the blue farm in the center. The numbers refer to the scenarios described in section 3.3.2, and scenario 4 refers to off-farm compost, described in section 3.3.1.

In scenario 2, composting is no longer a closed-loop system, because waste enters the system from elsewhere and is treated at the farm (i.e., food scraps from gardeners' homes or local restaurants). Here, the farm serves two functions: growing food, and treating waste. The additional function of avoided municipal waste treatment of biomass brought to the farm should be included. Allocation between these two functions is challenging because amounts of organic waste brought to the farm to be composted usually cannot be measured and tracked separately from organic waste generated on-farm. Then, the additional waste-treatment function should be accounted for using system expansion and substitution, by subtracting impacts of the alternate fate –incineration or landfilling– of organic waste from the UA system. This results in environmental credits to the UA system. This type of scenario is demonstrated in our case studies (Dorr et al., 2022b). Avoided fertilizer production should not be considered since the composted waste is used internally at the farm, so the benefit is accounted for in the LCA results by showing smaller impacts than if the farm had used fertilizer.

Scenario 3 composting can be found at urban farms that create inedible biomass waste (all farms) but do not use compost, such as soilless hydroponics or aeroponics systems. This type of composting represents a multifunctional process: it treats the farm's waste and creates a compost product. Here the UA site is a waste generator, as discussed in the off-farm compost section (section 3.3.1). Farms should be credited with avoided environmental burdens from production of the fertilizer or potting soil that the produced compost substitutes (Corcelli et al., 2019; Goldstein et al., 2016). Vieira and Matheus (2019) provide a comprehensive review and recommendations on the matter. Composting for waste treatment of biomass can account for 10-15% of climate change impacts in UA (Corcelli et al., 2019; Sanjuan-Delmás et al., 2018), but avoided burdens of fertilizer production can generate net impact abatements (Corcelli et al., 2019).

3.3.3 Carbon sequestration

Challenge:

Compost is rich in organic carbon that is stabilized and stored after application to soil (Lal et al., 2015). Carbon sequestration through composting with low-carbon soil management can remove carbon from the atmosphere (Tiefenbacher et al., 2021). From an LCA perspective, this represents avoided climate change impacts, where farms using compost should receive environmental credits for sequestering CO_2 as organic carbon in their soils. However, the long-term fate of organic carbon is mostly unknown and highly context dependent because of complex soil ecology. This introduces high uncertainty to a process that can largely influence LCA results (Mclaren, 2010; Strohbach et al., 2012; Tidåker et al., 2017). Existing agriculture soil carbon models are highly time and data intensive, and are poorly adapted to UA where unique substrate and high composting rates predominate (Dorr et al., 2017). .

Examples:

Several researchers argue for including carbon sequestration from compost in agricultural LCAs (Adewale et al., 2018; Martínez-Blanco et al., 2013) while others claim it is too poorly understood to be meaningfully considered (Joint Research Centre, Institute for Environment and Sustainability, 2012; Nordahl et al., 2022). Some compost LCAs (from a biowaste treatment perspective) have used carbon sequestration at rates of 10-14% of organic carbon (Boldrin et al., 2010; Tonini et al., 2020; Vaneeckhaute et al., 2018). Few UA LCAs have included soil carbon sequestration from compost. Dorr et al. (2017) used a soil model to estimate carbon sequestered from compost in substrate, potting soil, and amendments at an urban farm, , and concluded that sequestered carbon only offset 0.2-3% of GHG emissions of the farm. In a different UA LCA, Dorr et al. (2022b), applied standard carbon sequestration rates in a sensitivity analysis, which led to an offset of 3-23% of climate change impacts. LCAs of other urban green infrastructure, such as parks and golf courses, usually include carbon sequestration. This can largely affect results, sometimes even making the entire system a carbon sink (Bartlett and James, 2011; Nicese et al., 2021; Strohbach et al., 2012).

Recommendations:

We recommend excluding carbon sequestration from compost (or other organic inputs) in the main results of UA LCAs, due to the large uncertainties. It can be included in sensitivity analyses, or secondary results, to explore the extent to which it may be important, with care taken to highlight the uncertainty in those results.

3.3.4 Compost emission factors

Challenge:

The most impactful component of the compost life cycle is gaseous emissions of methane, nitrous oxide, ammonia, and volatile organic compounds during the composting process (Boldrin et al., 2009; Pergola et al., 2020). These emissions strongly affect climate change, acidification, eutrophication, and photochemical ozone formation impacts (Pergola et al., 2020). High variability in gaseous emissions from composting—due to differences in technical systems, feedstocks, and composting practices—result in high variability in composting impacts (Joint Research Centre, Institute for Environment and Sustainability, 2012).

Reference	Type of composting system	N_2O emissions	CH_4 emissions	GHG emissions	Notes (CO, NH_3, VOC emissions)
		(kg/ton fresh waste)			
Andersen 2010[a]	Home composting, closed unit	0.30-0.55	0.4-4.2	100-239	6 composting units
Martínez-Blanco 2010 HC[b]	Home composting bin	0.676	0.158	205.4	VOCs = 0.559, NH_3 = 0.842.
Martínez -Blanco 2010 IC[b]	Tunnel composting, with biofilters for fugitive gas	0.092	0.034	28.3	VOCs = 1.21, NH_3 = 0.11.
Colón 2010[c]	Fruit and vegetable scraps, yard waste, home composting	0.2	0.3	67.1	VOCs = 0.32, NH_3 = 0.03.
Quirós 2014 HE[d]	Home composting, high-emission system	1.16	1.35	379.4	Leftover fruits and veg, yard waste. NH_3 = 1.3.
Quirós 2014 LE[d]	Home composting, low-emission system	0.2	0.295	67.0	Leftover fruits and veg, yard waste. NH_3 = 0.03.
Ecoinvent v3.5[e]	Open windrow composting	0.025	1	32.5	Retrived from Ecoinvent.
AgriBalyse- GW[f]	Green waste	0.48	0.21	148.3	Green waste. VOCs = 0.14, NH_3 = 1.87
AgriBalyse- BW[f]	Bio waste	0.13	1.15	67.5	Biowaste. VOCs = 0.21, NH_3 = 6.23
Nordahl 2022 YW[g]	Yard waste, average from review	0.043	2.31	70.6	Average of 9 values
Nordahl 2022 OFMSW[g]	OFMSW, average, from review	0.068	0.879	42.2	Average of 21 and 19 values for CH_4 and N_2O
Nordahl 2022 manure[g]	Manure, average, from review	0.354	2.82	176.0	Average of 41 and 45 values for CH_4 and N_2O

Table 3 Emissions of N2O, CH4, and the sum of greenhouse gas (GHG) equivalents for N2O and CH4 are shown in kilograms of emission per ton of fresh waste composted, from some of the main sources of composting emission factors for urban agriculture life cycle assessments. GHG emissions are presented in kilograms of CO2 eq. OFMSW: organic fraction of municipal solid waste. a) Andersen et al., 2010, b) Martínez-Blanco et al., 2010, c) Colón et al., 2010, d) Quirós et al., 2014, e) Wernet et al., 2016, f) Asselin-Balençon et al., 2020, g) Nordahl et al., 2022.

Examples:

Many UA LCAs use composting emission factors from Andersen et al. (2012, 2011), Martínez-Blanco et al. (2010), and Colón et al. (2010), because they measured inventory data specifically for home composting, which is representative of small scale, on-farm composting operations. The LCA database Ecoinvent (Wernet et al., 2016), which uses inventory data from Edelmann and Schleiss (1999), is also commonly used to model composting in UA and conventional agricultural LCAs. Table 3 shows the wide range in composting GHG emission factors from sources commonly used in agricultural LCAs. This selective list of emission values highlights the potential pitfalls from selecting composting inventories with such variability. Indeed, in our case study we found that climate change impacts were reduced by 2-14% when we used the inventory from Ecoinvent rather than from a meta-analysis by Nordahl et al. (2022). For more complete summaries of measured composting emission factors, see reviews papers by Nordahl (2022), Boldrin (2010), and Amlinger (2008), and discussion section reviews in Quiros (2015) and Avadi (2020).

Recommendations:

To address variability in composting emission factors for UA LCAs, we recommend modeling multiple scenarios with different emission factors when a farm applies large amounts of compost. Emission factors can be chosen from a specific source with a representative composting technology, or averages of multiple sources can be used. Monte Carlo simulations can be performed to include a distribution of composting emission factors to obtain a range of results and check for changes in directionality when comparing between systems (e.g., UA against conventional agriculture).

3.4 Substrate

Challenge:

A unique characteristic of UA compared to rural agriculture is that it is not necessarily carried out on soil. Soil, or top-soil, is defined as natural bodies made of organic and inorganic material that are formed at the surface as the result of complex biogeochemical and physical processes (Brevik and Arnold, 2015; Hartemink, 2016). Using soil as a growing medium is often not an option in UA due to soil pollution in cities, or lack of greenfields. In these cases, soilless cultivation methods are used (such as hydroponics, aeroponics, or aquaponics), or a substrate/growing medium is imported. In an LCA, substrate can be considered infrastructure that requires material inputs of large quantities and variable types. Current practices around producing substrate in UA LCAs are unclear, because it often goes unmentioned, it seems to be inconsistently included, and system modeling decisions around the recycled materials often incorporated in substrate are variable (Dorr et al., 2021a). Yet, several UA LCAs have found that creating and replenishing substrate was the largest contributor for most impact categories (Dorr et al., 2017; Kim et al., 2018; Vacek et al., 2017).

As a fixed input/infrastructure, substrate's lifetime directly affects its impacts, but very little information is available regarding the expected or actual lifetimes of substrate in UA. Since substrate is often amended, replenished, and used indefinitely, its lifetime is probably not limited by the material itself. Rather, substrate lifetime will likely be determined by the lifetime of the UA project or the building it is located on (Romanovska, 2019). There are few records of the lifetime of UA projects, but given UA's sometimes transient or uncertain economic nature, we suspect that such lifetimes may be shorter than anticipated (Demailly and Darly, 2017).

Examples:

Peat, coir, wood and compost are commonly used to produce substrate (Barrett et al., 2016). In UA, materials such as crushed brick, spent coffee grounds, spent brewer's grain, and shredded paper have also been observed (Dorr et al., 2021b; Grard et al., 2020; Martin et al., 2019). The numerous possible substrate inputs, mostly co- or by-products, lead to many options for modeling the materials.

Limited details are available regarding lifetime and fate of permanent substrates in UA LCAs. Dorr et al. (2017) evaluated a research-oriented rooftop farm using substrate in raised beds, and assumed a 10-year farm lifetime and that substrate had no end-of-life treatment (as it would be donated and reused). Kim et al. (2018) evaluated a rooftop farm and green roof, and assumed a 40-year lifetime based on the durability of the roof membrane material. Vacek et al. (2017) did an LCA of green roofs and assumed a lifetime of 20 years, noting that they would require renovation after this point. They assumed that substrate would be landfilled, being too degraded for recycling/reuse.

Recommendations:

Regarding materials, the LCA guidelines published by Growing Media Europe (2021) detail how to model and what to include for numerous substrates found in UA. Peat and peat moss have been well studied, and the processes available in LCA databases should be used. Impacts for coconut and wood/bark-based materials should be allocated on an economic basis between the main coconut and forestry products and the substrate byproducts (European Commission, 2010b). Residual waste products that have negligible economic value should not incur impacts from the first use, according to economic allocation principles (ISO, 2006b). For both valuable byproducts and residual waste products, impacts from their transport after the original site of use, and energy and water needed for processing into substrate should be accounted for (Growing Media Europe, 2021).

For permanent UA substrates (i.e. not disposable substrate in hydroponics and aeroponics), impacts from the initial installation of substrate should be allocated over the lifetime of the farm, similar to other pieces of infrastructure. This lifetime is usually highly uncertain, but a timeframe of 10-40 years can be considered, which can be refined based on the orientation and precarity of the case study. Results can be sensitive to this assumption so sensitivity analyses should evaluate scenarios with different farm/substrate lifetimes. Disposable substrate used in hydroponics and aeroponics do not have the same lifetime considerations and can be treated as a supply.

Replenishing substrate helps maintain substrate volume and quality. Impacts of these replenishments should be temporally allocated to the time between applications. For example, if substrate is replenished every two years, then half of the amount applied can be allocated the system in an LCA considering one year of production.

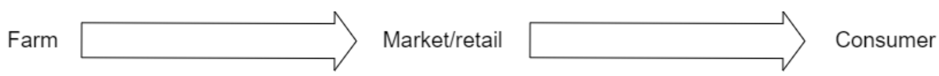
End-of-life for inorganic substrates will likely include municipal waste treatment or recycling. Organic substrates are mostly composted or applied to fields as a soil improver (Growing Media Europe, 2021). For composting, the farm can be seen as the waste-generator described in section 3.3.4, and impacts of composting should be allocated between the waste-generator and the compost user. If substrate is applied as a soil improver by the next user, and no treatment or processing are necessary, then no impacts for waste treatment should be given to the farm.

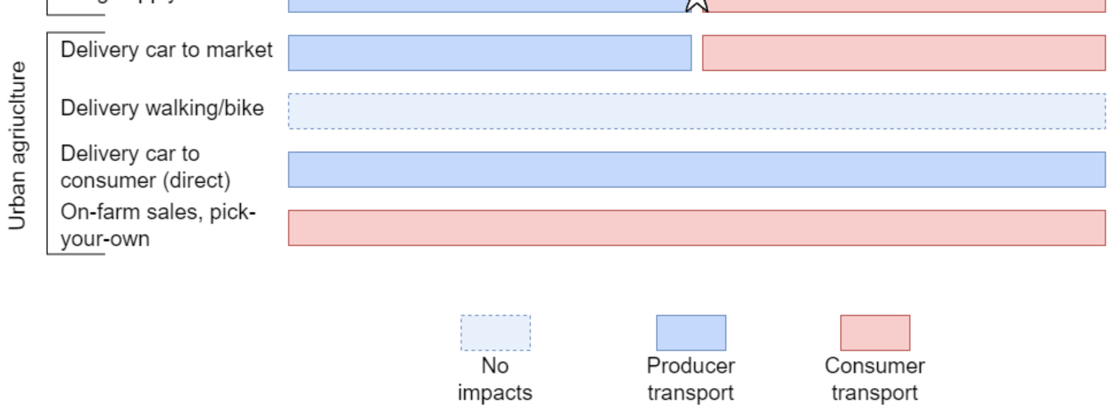
Increased transparency and improved reporting regarding substrates are necessary in UA LCAs. The nature and the origins of substrate material should be clearly described, plus physio-chemical characteristics, if available (Barrett et al., 2016). The amount of substrate initially applied, the amount added in amendments, the lifetime, and end of life waste treatment should be clearly stated.

3.5 Transport and delivery

Challenge:

A main supposed environmental benefit of UA is that it limits food-miles through proximity of producers and consumers (Kulak et al., 2013; Weidner et al., 2019). Yet, knowledge is scarce about how UA products are transported/delivered—yet alone their environmental performance. This benefit is sometimes dismissed, considering that on average, transportation accounts for 6-11% of climate change impacts from food systems (Poore and Nemecek, 2018; Weber and Matthews, 2008). However, fruits and vegetables can have larger contributions to climate change impacts from transport (often 10-25%, but as high as 54%), due to the potential relatively lower impacts at the farm-stage, long distances, refrigerated transport, and airplane travel (Barbier et al., 2019; Bell and Horvath, 2020; Poore and Nemecek, 2018; Weber and Matthews, 2008). The benefit of reduced transport is mostly tested through comparisons of UA LCA results to the supply chains of rural agriculture (Dorr et al., 2021a).





products from UA are delivered by walking or by bike, because there are almost no impacts

Figure 3 The downstream system boundary is shown for several simplified distribution schemes. Colored bars indicate who is doing the travel, and the empty bar for walking/bike indicates that there are no environmental impacts from this travel. Many rural food LCAs have a system boundary that ends at the market/retail stage (shown by the star). Several types of distribution networks that are common for UA are inconsistent with this system boundary, because there is no equivalent market/retail stage. UA LCA practitioners should ensure consistent system boundaries when comparing results with rural food LCAs.

to/by the consumer, also called the 'last mile', is usually not included in food LCAs, and the system boundary ends at the market/retail stage (the star in Figure 3) (Pérez-Neira and Grollmus-Venegas, 2018). It is usually not included because it is difficult to model consumer transport behavior, and to isolate transport specifically for food purchases from other transport. Therefore, many comparisons between urban and rural agriculture products risk comparing a cradle-to-consumer UA system with cradle-to-market rural agriculture system. The last mile step is important because it can account for an even larger food transport distance (or 'food miles') than the transport over long food supply chains (Majewski et al., 2020). Impacts of the last mile for food (from customer travel) can contribute up to 21% of life cycle climate change impacts of pasta (Gnielka and Menzel, 2021), or 6% of urban food system climate change impacts (Stelwagen et al., 2021). Therefore, inaccurate comparisons here may omit a large benefit of reduced consumer transport of UA.

Examples:

Transport from the farm to the consumer on foot or by bike, or when production occurs in or on a building where consumers live or work, has been considered in several UA LCAs. They state that there are no processes or impacts for delivery (Figure 3) (Sanjuan-Delmás et al., 2018; Sanyé-Mengual et al., 2018a; Torres Pineda et al., 2020). Several UA LCAs include distribution by car to the consumer, based on a simplified model/distribution of transport modes and distances from the distribution point to consumers' homes (Hall et al., 2014; Hu et al., 2019; Pérez-Neira and Grollmus-Venegas, 2018). Other LCAs regarding urban food consumption and food products have focused on the last-mile transport impacts (Bevilacqua et al., 2007; Melkonyan et al., 2020; Stelwagen et al., 2021).

Recommendations:

UA LCAs should include post-farm delivery processes to account for the unique urban position of farms (Weidner et al., 2019). Since there may be large uncertainties in delivery logistics and inconsistent system boundaries with rural systems, results should be presented with and without post-farm transport, giving cradle-to-farm-gate and cradle-to-consumer or -market impacts (Sanyé-Mengual et al., 2015a). This is particularly relevant for comparisons to rural agriculture because proximity to the consumer is a core characteristic and environmental benefit of UA. The delivery scheme of a case study should be clearly described, including the transport distances, modes, and frequencies of deliveries.

When comparing urban and rural agriculture, careful consideration must be taken to ensure that system boundaries are consistent. In particular, if the UA system has no impacts from transport, because it is done on foot or by bike, then the impacts are the same with a cradle-to-farm gate or cradle-to-consumer boundary. A cradle-to-consumer boundary is implied and should be considered, for comprehensiveness and to account for this environmental benefit of UA. Then, a scope including transport to the consumer should be included for the rural system. This stage is not represented in food products in LCA databases, and several additional transport steps are necessary for the product to reach the consumer. The feasibility of this is uncertain, however, given the lack of last mile transport data.

Due to the difficulty of modeling these complex distribution networks, in-depth research on this topic may need to be done separately from production-focused UA LCAs (Coley et al., 2009; Stelwagen et al., 2021). This represents an opportunity for cross-disciplinary research on UA production and urban mobility. A city or foodshed scale may provide additional insight, as this topic quickly veers into the larger urban food logistics system rather than farm systems (Benis and Ferrão, 2017; Melkonyan et al., 2020).

3.6 Variability and uncertainty of UA

Challenge:

Agricultural LCAs have particular issues with high variability because of diversity in controlled factors like farming practices and logistics, and in ‘natural’ factors like climate and soil characteristics (Lam et al., 2021; Mclaren, 2010; Notarnicola et al., 2017). We hypothesize that the controlled factors are even more variable in UA than in rural agriculture. Urban settings introduce physical limitations (i.e. shading from buildings, poor-quality anthropogenic soils, air pollution, and limited access to materials) which spur diverse growing practices and setups (Taylor, 2020; Wagstaff and Wortman, 2015). Human elements such as motivation for urban farming, years of experience, and access to agronomic information and training are highly variable, and likely affect growing practices (McClintock et al., 2016; Taylor, 2020). More broadly, the novel and semi-formal status of much of UA means that it has not converged towards optimized, standardized operations. In contrast, rural agriculture has been researched for decades, practiced for thousands of years, and is relatively consistent due to farmer trainings, university agricultural extension support, and technology such as tractors, crop varieties, and chemical inputs (Armanda et al., 2019; O’Sullivan et al., 2019).

These factors lead to variability at a given farm (i.e. within systems). This can manifest as practices changing throughout the year, or spaces across the site being managed inconsistently. Uncertainty is also problematic, since many data are likely unavailable. This poses a problem for studying a system in its representative, average, ‘steady’ state. It also challenges the common LCA practice of substituting unavailable primary data with secondary data, based on the assumption that systems have somewhat standard and predictable practices.

There is also high variability in UA overall (i.e. between systems). Indeed, in the review of UA LCAs (Dorr et al., 2021a), there were few actual replicates of systems due to diverse growing technology, motivation, climate, and others factors, making it difficult to compare results. Plus, many case studies were research-oriented or used innovative practices, suggesting that they may not have been representative systems. This poses a challenge to understanding the general performance of UA, since there is not really a 'general' situation for UA.

Examples:

One of the most common ways of addressing variability and uncertainty in UA LCAs is presenting alternative scenarios in the form of sensitivity analyses. This was done in UA LCAs by modeling different infrastructure lifetimes (Dorr et al., 2017; Martin and Molin, 2019), crop yields (Romeo et al., 2018; Rufí-Salís et al., 2020b), or light efficiency for indoor systems (Pennisi et al., 2019; Shiina et al., 2011). Another strategy was to use ranges of inventory values, for example for delivery/distribution schemes, generating a range of results (Hu et al., 2019; Pérez-Neira and Grollmus-Venegas, 2018; Stelwagen et al., 2021). When parameters with high variability are identified, the goal of the LCA can shift to find tipping points where one system performs better/worse than another (usually UA vs rural). This was done for yield and distance from producer to consumer (Kulak et al., 2013; Sanyé-Mengual et al., 2015a). Alternatively, Monte Carlo simulations can be employed to quantify ranges of results based on distribution of parameters, such as composting emission factors and bulk density (Dorr et al., 2022b).

Recommendations:

Variability and uncertainty within systems can be reduced or accounted for with several strategies. Temporal variability, due to annual climate differences or changes in operations (for example due to farmer turnover), should be reduced by collecting data for multiple years and using an average of values, or selecting the most representative year (Loiseau et al., 2020). Specialized indicators can be used that quantify how important variability is for a system (Hauck et al., 2014). Variability of inventory items should be considered using distributions or ranges (Stelwagen et al., 2021), and probabilistic simulations, such as Monte Carlo simulations (Huijbregts, 1998).

Variability between systems is problematic when trying to compare or summarize results for similar systems. Such comparisons are necessary to draw trends and generalize LCA findings, which is a feature of rather mature LCA research topics. Few technical recommendations can be made here, but we note that more complete, transparent descriptions of case studies would help readers interpret results and make more relevant, accurate comparisons between studies.

4 Research directions for UA LCAs

This section presents aspects of UA LCAs that should be the subject of future research. These topics should not necessarily be systematically included in UA LCAs, because more research and development are needed. Still, we present practical recommendations for including them in UA LCAs now. We discuss research directions that can improve UA LCAs, and how applying LCA to UA can lead to insights for LCA overall.

4.1 Align with urban land uses and green infrastructure LCAs

Presentation:

The UA LCA literature is dominated by a product-based perspective, which inherently places the focus on the food-production function of UA. UA distinguishes both the unique, non-rural position of agriculture, plus the non-conventional use of urban space (Neilson and Rickards, 2017). The latter perspective has not been widely studied with LCA, except for studies comparing different uses of rooftops for flower gardening, farming, or solar panels (Corcelli et al., 2019; Goldstein et al., 2016; Kim et al., 2018). UA is one option for urban green infrastructure among many others, and may be more comparable to a park or other social/recreational activity than it is to rural agriculture. There is a wealth of literature on environmental assessments of urban parks and forests (Strohbach et al., 2012), golf courses (Tidåker et al., 2017), urban wetlands (Duan et al., 2011), grassy areas (Smetana and Crittenden, 2014), and other green infrastructure (Nicese et al., 2021), and it would be useful to relate UA to these land uses. It could provide meaningful comparisons to similar systems, and illuminate shortcomings in UA LCAs that are obscured by a product-based perspective. For example, urban green infrastructure LCAs found that waste treatment of biomass can be highly impactful (Nicese et al., 2021; Tidåker et al., 2017), and results can be highly sensitive to carbon sequestration (Strohbach et al., 2012; Tidåker et al., 2017), which has not emerged in UA LCAs.

Recommendations:

We call for increased attention to this unexplored research direction for UA LCAs: adopting an urban green infrastructure perspective of UA. Here, UA is seen as multifunctional with land use/green infrastructure as the main function, and food production is a secondary function that should be dealt with through allocation or system expansion. With system expansion, the impacts of producing an equivalent amount of food could be subtracted from the farm's impacts. With allocation, the repartition of revenue from food sales compared to grants or other sources of funding could be used for economic allocation.

4.2 City-scale/Scaling up

Presentation:

UA is often presented as a tool for sustainable cities (Petit-Boix et al., 2017). Evaluating the effects of UA on resource consumption, food provisioning, and environmental impacts at the city-scale is useful to determine the relative magnitude of findings from the farm-scale. It can also identify emergent processes at the city-scale that are not evident at the farm-scale, such as effects on municipal organic waste treatment or urban transport logistics.

The effects on the city of "scaling up" or developing UA has been modeled under different scenarios. Goldstein et al. (2017) evaluated the effect of installing UA in available land in Boston, USA, and found that it could reduce food-related climate change impacts at the city level by 1-3%, and increase land occupation by 1%. Mohareb et al. (2018) performed a similar analysis for the USA and found food sector GHG emissions decreased by 1%. Other scaling-up analyses suggest that UA could 'absorb' and compost 9% of municipal organic waste in Boston (Goldstein et al., 2017), 17% in Lyon, France, and 52% in Glasgow, Scotland (Weidner and Yang, 2020). Extrapolating farm-level results to the city-scale helps provide perspective, because if for example fruits and vegetables are substantially more or less impactful than rural products, but at the city or individual diet scale the difference is meagre, then UA LCAs should shift framing away from UA being a tool for reducing impacts of urban food consumption. Such research requires estimates of the current diets of city inhabitants (to evaluate substitution effects) (Dorr et al., 2022a), the available space for UA (Saha and Eckelman, 2017), and current city-scale flows of materials such as water and organic waste (Weidner and Yang, 2020).

UA is embedded in the infrastructure and functioning of specific cities, which provide certain environmental constraints or opportunities based on the city context (Martin et al., 2016). For UA LCAs, some characteristics of the specific city are inextricably included in the LCA results. For example, a well-known factor at the country level is the electricity grid (Dorr et al., 2021a). Similar factors at the city-level may influence UA environmental performance, such as city density, which may determine the proportion of rooftop vs ground-based UA, or the transport mode for product delivery (Montealegre et al., 2021). The building stock in a city may affect UA's form and impacts: for example, older buildings are more likely to need structural reinforcement for rooftop UA (Ledesma et al., 2020). The typical waste treatment scheme for organic waste in a city would largely influence the potential for avoided burdens related to compost—i.e. if organic waste is composted anyway through the city. Finally, the benefits of reduced food miles for rural products are context-specific, and depend on the actual source and distribution network of products to a city (Bell and Horvath, 2020; Edwards-Jones et al., 2008; Hospido et al., 2009).

Recommendations:

We recommend that researchers apply LCA to UA at the city scale, which can put farm-level impacts and benefits into perspective, and account for context-specific aspects of UA in a given city. As this scope veers away from on-farm production, and may focus on other aspects such as transport and delivery or external consequences of UA, primary data from farms may be less essential. Farm-level LCAs should include descriptions of the city to facilitate interpretation by others, such as the position of the farm in relation to the city center/boundary, city density, and the role of UA in the city (i.e. its history, orientation…).

4.3 Ecosystem services and positive impacts

Presentation:

LCA is designed to evaluate the negative (adverse) impacts of a system rather than its positive impacts (benefits). The ecosystem service (ES) concept takes the opposite perspective, defined as the benefits that people obtain from ecosystems (Millennium Ecosystem Assessment, 2005). ES assessments may better measure the benefits of UA than LCA, and combining the two ways of thinking would allow for more comprehensive assessments of UA. There is no consensus on how best to measure ES, although there are many methods available (Grêt-Regamey et al. (2017) evaluated 68 of them). Much work has been dedicated to the consideration of ES in LCA (Maia de Souza et al., 2018; Othoniel et al., 2016; Tang et al., 2018; Zhang et al., 2010), although no method is consistently used. Some rural agriculture LCAs have performed allocation using ES (Boone et al., 2019) or with ES modeling (Chaplin-Kramer et al., 2017), but no UA LCAs have incorporated ES. ES may be fully integrated into the LCA methodology (i.e. with additional impact pathways for LCA,), or may be more loosely integrated though qualitative or quantitative interpretation of results calculated separately from an LCA (De Luca Peña et al., 2022).

UA is a particularly rich topic through which to promote methodological development of ES and LCA. It would offer useful case studies for future research because ES have been widely measured as a benefit of UA, both qualitatively through interviews with stakeholders and ranking of ES (Camps-Calvet et al., 2016; Sanyé-Mengual et al., 2020) or quantitatively with indicators (Cabral et al., 2017; Grard et al., 2018).

There are four types of ES: provisioning, regulating, cultural and supporting (Millennium Ecosystem Assessment, 2005). Food production in UA is an obvious provisioning service. As many UA LCAs use a FU based on food production, they essentially quantify the impact of this ES. Boone et al. (2019) demonstrated a method to allocate between this provisioning ES of agriculture and other ES in an LCA, which highlighted that food was not the only ES (or ‘output’) of agriculture.

Regulating ES of UA that have been measured include water runoff regulation, organic waste recycling, and microclimate regulation (Dennis and James, 2017; Grard et al., 2018). Benefits of avoided stormwater runoff have been quantified with LCA, and offset 13-72% of several impact categories (Goldstein et al., 2016; Kim et al., 2018). Carbon sequestration can also be evaluated using LCA or ES (Orsini et al., 2014), and its implication in LCA is described in section 3.3.3. Reduction of the urban heat island effect is a frequently proposed regulation ES of UA, and is generally excluded from all LCAs (Susca and Pomponi, 2020).

Cultural ES are sometimes perceived as the top benefit of UA, and include recreation, beautification, cultural identity, social cohesion, community building, and education (Giacchè et al., 2021; Sanyé-Mengual et al., 2018b). Indicators to measure cultural ES include the volunteer hours, number of educational and recreational activities offered, and their number of participants (Dennis and James, 2017; Giacchè et al., 2021). Cultural ES may provide a framework to include social benefits in UA LCA assessment (detailed more in section 4.4).

The role of biodiversity in ES is foundational, as it is defined as the source of ES (McDonald et al., 2013; Millennium Ecosystem Assessment, 2005), and is often used as a proxy indicator for supporting ES (Cabral et al., 2017). Improved local biodiversity is perceived as an important environmental benefit of UA (Sanyé-Mengual et al., 2018b) and is frequently measured in the context of ES of UA (Dennis and James, 2017; Quistberg et al., 2016). This benefit is not accounted for in LCA. Biodiversity impacts in LCA have been the subject of methodological development for decades, and is usually framed as the impact *on* biodiversity *from* land use (or other ecological damage, although most frequently land use) (Teixeira et al., 2016). LCA models the upstream and downstream impacts of materials and processes on biodiversity around the world, and does not consider local biodiversity (Teixeira et al., 2016). Other measures are more relevant for farm-scale biodiversity impacts like species richness, habitat fragmentation, habitat vulnerability, or land use intensity indicators (Frischknecht et al., 2016; Pepin, 2022).

Recommendations:

For practitioners looking to operationalize ES and LCA for UA, results from each method can be qualitatively assessed in parallel or quantitatively through composite indicators (De Luca Peña et al., 2022). For an integrated assessment, for example comparing types of UA within one study, results can be integrated in a multi-criteria decision analysis (Ledesma et al., 2020).

Researchers looking to improve LCA methodology by integrating it with ES should consider using UA as their application. UA represents a particularly relevant activity, due to its multifunctionality and the fact that many ES have already been demonstrated.

4.4 Social benefits and life cycle sustainability assessment

Presentation:

A main strength of UA is its multifunctionality, with important social functions (Gomez Villarino et al., 2021; Orsini et al., 2020). This is rarely reflected in UA LCAs, but it should be, since core principles of LCA are evaluating the main function of a system (through selection of a FU), and accounting for multiple outputs (through allocation and system expansion).

Accounting for social aspects of an activity is a main issue for LCA, and social LCA (S-LCA) is a promising yet nascent strategy to overcome this (UNEP/SETAC, 2009; Zimek et al., 2019). Using life-cycle thinking, S-LCA tracks the social impacts of a product’s life cycle. S-LCA quantifies negative impacts, and therefore may not be appropriate for evaluating the social benefits of UA. S-LCA databases offer data for social impacts embedded along the supply chain, but the information necessary for UA is more relevant at the farm, neighborhood, or city scale (Romanovska, 2019). Plus, such databases are not as generalizable as large LCA databases. A strength of S-LCA is its ability to account for the perspectives of multiple stakeholders, such as workers, consumers, and the local community. This is especially useful to evaluate the potential for UA to address social justice issues, by highlighting not just which social benefits are brought, but who they are affecting. S-LCA currently lacks agreed upon social indicators, partly because they are situational and defined through stakeholder engagement, making consistent methods and comparisons between studies difficult (Fauzi et al., 2019). Peri et al. (2010) outlined indicators for S-LCA of green roofs, including area of green roof made accessible to the public, fair salary, working hours, air pollutant levels, and outside air temperature.

Apart from S-LCA, an option to include social benefits of UA is to address its multifunctionality with traditional LCA practices. For example, allocation can be used to distribute impacts based on relative importance of food production vs. social benefits. This allocation may be done based on the level of ES provided by each activity, as done in Boone et al. (2019). Alternatively, it may be based on the relative sources of revenue from food sales vs. grants vs. other activities. If social goals are the main function of a farm, we can imagine using a FU based on the social “output”, such as volunteer hours or total number of new people met by UA participants, which can be linked to cultural ES.

Social aspects of UA may be evaluated in parallel to environmental impacts from LCA rather than being fully integrated into LCA. Indeed, many researchers acknowledge that LCA cannot capture everything, and it is useful to complement it with other methods (De Luca Peña et al., 2022; Fauzi et al., 2019). In practice, this would be most useful to compare different types of UA within a study, where the same data can be collected from a set of urban farms. UA LCA practitioners should strive to measure these indicators of social benefits and present them in case studies, even when they are not based on a life-cycle approach.

The LCA community has promoted and strives for life cycle sustainability assessment, which combines environmental LCA, life cycle cost analysis (LCCA, which was reviewed for UA by Peña and Rovera-Val (2020)), and S-LCA. Such holistic life cycle sustainability assessments are still largely more aspirational than operational (Fauzi et al., 2019; Finkbeiner et al., 2010). We urge UA LCA practitioners to consider measures of economic and social sustainability even if they are not life-cycle based, which is indeed particularly data-demanding (Sanyé-Mengual et al., 2017). LCA results may even be included in broader indicator-based sustainability assessments, which are operationalized in tools for rural agriculture and are under development for UA (Clerino and Fargue-Lelièvre, 2020; Hély and Antoni, 2019).

Recommendations:

Researchers should work towards defining a set of S-LCA indicators relevant for UA. The concept and assessment of cultural ES may serve as a basis here, since they are both indicator-based, site-specific measures. New methods should be tested to use allocation or alternative, social-based FUs to account for social aspects of UA. Although we should ultimately strive for life cycle sustainability assessment, non-life cycle indicators and results, such as results from surveys and interviews, should be presented alongside LCA results to provide more holistic views of sustainability.

5 Conclusion

Since the first LCA of UA a decade ago, interest in and knowledge of the environmental performance of UA has increased. Still, large uncertainties remain regarding best practices for these assessments, and even defining what questions we aim to address. In this article, we laid out recommendations and research directions that are intended to improve LCAs of UA. These improvements can lead to more thorough LCAs and more consistency between case studies. We also outlined the questions that UA LCAs may aim to answer, in the hopes of bringing perspective and clarity to this field of research. Finally, this work highlights what LCA can *learn* from UA through challenges in applying it to this complex and multifunctional activity. To accurately support policy and decision-making around UA, LCAs must be more comprehensive. To provide more meaningful support, UA LCA findings should be considered alongside measurements of other sustainability dimensions, whether they are life-cycle based or not.

By applying these guidelines and strengthening UA LCAs, this research topic can better support environmental sustainability of UA and cities. This research can better inform policy makers about how UA implementation will affect environmental performance of cities, and which types or characteristics of UA to leverage for specific goals. It can inform urban farmers on how to operate or design their farms to minimize environmental impacts. They can better understand which changes to implement, and which ones may not be worth the effort given small environmental gains. Finally, the research community can explore methods to enhance the use of LCA for multifunctional, complex activities, such as UA.

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