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

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ABSTRACT

There is increasing interest in evaluating the environmental performance of urban agriculture (UA), primarily 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. This article outlines the key challenges of applying LCA to UA and recommends concrete steps to bring consistency and comprehensiveness to the topic. The research questions that LCA can address are framed before providing 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, the article proposes future research areas that will benefit LCA generally and its application to UA, such as framing UA as urban green infrastructure, evaluating 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.

1. Introduction

Urban agriculture (UA) is a multifunctional activity with many assumed and demonstrated benefits for cities and residents. These potential benefits from a social, economic, and environmental standpoint position UA as a powerful tool to improve urban environments, contribute to more sustainable urban food systems, and enhance the well-being 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 and greenhouse gas (GHG) emissions (Nicholls et al., 2020; O'Sullivan et al., 2019). Holistic environmental evaluation methods, such as life cycle assessment (LCA), are needed to capture impacts across the food value chain. Although LCA is standardized, its outcomes for UA systems are highly variable because of inconsistencies in how LCA is performed (Dorr et al., 2021a). Thus, reliable answers to crucial questions surrounding the environmental performance of UA are lacking. 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 food production? Researchers require guidance to more consistently make decisions regarding system modeling, system boundaries, and reporting so that LCAs of UA can help answer such questions. General frameworks and guides have been proposed to improve the rigor and comparability of LCAs, such as the Product Environmental Footprint Category Rules Guidance (European Commission, 2017), while others have targeted specific sectors facing unique methodological issues. 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 that caused a reappraisal of government policies to support first-generation biofuels (Searchinger et al., 2008).

To avoid similar mistakes, 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),

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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). However, none of the above articles address the particular context of urban farming, which warrants specific analyses and recommendations.

This study intends to fill this gap by providing a guideline specific to the assessment of UA with LCA. It is applicable to all UA forms, the 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., 2023a).

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 unique attributes of UA in LCA, reviews how these attributes are currently considered 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 LCAs of conventional agriculture and UA are included here. By outlining clear rules for dealing with the unique challenges of applying LCA to UA, future work can be done in 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. Methodology

This article relies primarily on analyzing a set of UA LCAs retrieved through a systematic literature review done in April 2021 (Dorr et al., 2021a). Briefly, references were searched in the online databases Web of Science and Scopus using the key terms "urban agriculture" and "life cycle assessment," along with some variants (e.g., urban garden, farm, greenhouse, or plant factory). References were checked and screened regarding focus and compliance with LCA standards. LCA practices were surveyed in this set of UA studies, particularly the goal and scope, system modeling, and inventory phases. They were benchmarked against more generic recommendations in articles dealing with the methodology of LCA application to agriculture and food systems, retrieved from another systematic literature search. The recommendations established from this survey of current practices and possible improvements were further tested with a set of novel UA case studies on eight urban farms and community gardens in France and California (Dorr et al., 2023a). The lessons learned from applying LCA to these cases, using an initial version of the framework presented here, were used to improve it and validate its relevance. Recommendations regarding the improvement of farming practices were also tested with some of the farmers and farm managers.

3. Why do LCAs of urban agriculture?

Since there are diverse framings of UA, it is useful to clarify how it should be assessed with LCA by defining both the goals and the larger questions LCA aims 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 multiple dimensions of UA. Table 1 highlights some key, largely unanswered questions around UA that LCA can address. Some of these are already prevalent in the literature, while others are original suggestions which have yet to be addressed. 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 functional unit recommendations, is in Appendix A.

Table 1
Lists the broad questions which LCA of UA can address in principle, along with their description/justification and possible functional units (FU) in brackets. See Appendix A for concrete examples of previous studies, questions addressed and FUs.

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, it is worthwhile to decide whether UA is a useful strategy.	Single or mixed crops (kg fresh matter); cost/revenue (€/$), individual diet (cal), city-wide food flows (tonnes of material)
Is UA an environmentally positive type of green infrastructure to implement in a city?	Green infrastructure is promoted in cities, and many types are possible. City leaders must decide which types to implement.	Land area (ha); cost/revenue
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 (capita.yr); land area; operation of other sectors (e.g., waste treatment; tonnes of material processed)
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 trade-offs between the social and environmental dimensions? Can the social benefits of an activity justify its adverse impacts on the environment?	Single or mixed crops; land area; cost/revenue; total operations of urban farm (FU: one farm); social functions (e.g., hours of education, number of participants as FU)
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 or mixed crops, land area, cost/revenue, total operations of urban farm, social functions (e.g., 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, practitioners should be informed of the best practices to minimize their impact.	Any

4. Challenges and practical recommendations for UA LCAs

This section describes the unique aspects of UA that present methodological challenges for LCAs alongside recommendations to address 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 4.3, on compost, includes additional subsections because there are numerous challenges, and the methodological issues associated with the inclusion of compost inputs in agricultural LCAs has not been reviewed. 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 5.

4.1. Crop diversity

4.1.1. 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 principles 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; Pourias et al., 2016). It is unreasonable to expect urban farmers to distinguish inputs for so many crops, thus LCA practitioners often contend with the challenge of including many crops in one FU.

Table 2
Key recommendations for performing UA LCAs according to their position along the 4-step LCA process.

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 transportation 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.

4.1.2. Examples

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). 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 complete a life cycle inventory for each crop (Caputo et al., 2020; 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). Others have used a simplified FU covering a basket of crops (e.g., 1 kg 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 an FU of kilogram of mixed crop (Christensen et al., 2018; Pepin, 2022), which complicates interpretation due to differences in nutritional value.

4.1.3. Recommendations

The main options for dealing with multi-crop systems are to evaluate a basket of products (by mass or by converting to calories or nutritional indexes), allocate between products, or choose an FU that is not based on food production. It is impossible to universally recommend an 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 some best practices can be recommended. When an 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 an FU of production of a single crop, but due to high crop diversity this may not be feasible. Finally, providing results across multiple FUs can illuminate tradeoffs and compensate for the opaque nature of mixed-product FUs.

4.2. Data availability

4.2.1. 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 (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., 2023b). 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).

4.2.2. 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).

4.2.3. 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 (e.g., 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 containers emptied times container volume (Pollard et al., 2018). Researchers should periodically check for leaks in irrigation systems, which can be substantial (Dorr et al., 2023b). 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., 2023a) provides concrete examples of how to collect data across diverse systems.

4.3. Compost

Compost is the main input to many urban farms (see detailed review in Appendix B) (Dobson et al., 2021; Edmondson et al., 2014). An 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; Martínez-Blanco et al., 2009) and for UA (Dorr et al., 2023a; Liang et al., 2019). Therefore, compost is given extra attention for this section.

4.3.1. Off-farm compost system modeling

4.3.1.1. 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; Toniolo et al., 2017).

4.3.1.2. 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 transportation 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 Product Environmental Footprint (PEF) method of the EU recommends this allocation method using a complex formula (the Circular Footprint Formula) involving a factor sharing the burden between the waste producer and re-user (Zampori and Pant, 2019). 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) (Guinée et al., 2004). This can be 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).

4.3.1.3. Recommendations. Off-farm compost should be treated 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 (Figs. 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).

4.3.2. On-farm compost system modeling

4.3.2.1. Challenge. On-farm compost refers to the composting

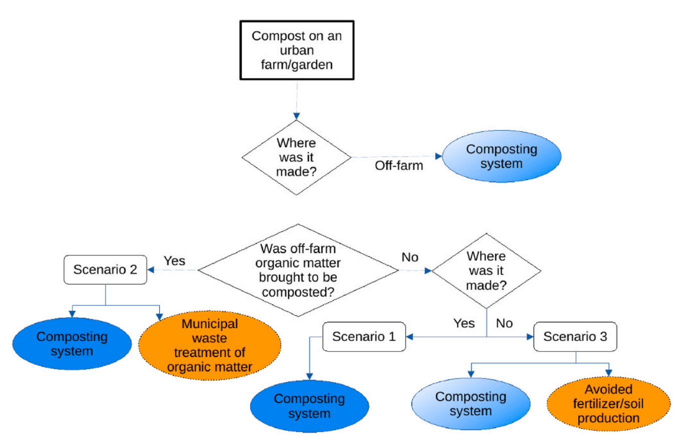

Fig. 1. Decision tree providing guidance on the handling of composting for an urban farm. Blue circles represent impacts from composting emissions and orange circles represent substituted processes that can be subtracted from the farm's impacts thanks to composting. Blue circles with gradients represent 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 4.3.2. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

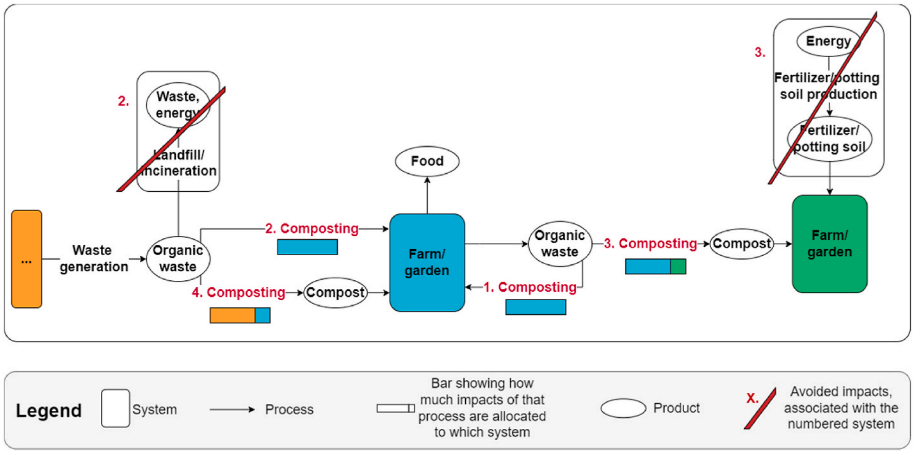

Fig. 2. Process diagram presenting the different composting scenarios as described in *section 4* from the perspective of the farm in the blue box. The numbers refer to the scenarios described in section 4.3.2, and scenario 4 refers to off-farm composting, described in section 4.3.1. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

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

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 (e.g., 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.

4.3.2.2. 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, 2006).

In scenario 2, composting is no longer a closed-loop system because waste enters the system from elsewhere and is treated on-farm. 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 (Dorr et al., 2023a). 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 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 4.3.1). 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 GHG emissions in UA (Corcelli et al., 2019; Sanjuan-Delmás et al., 2018), but avoided burdens of fertilizer production can generate net GHG reductions (Corcelli et al., 2019).

4.3.3. Carbon sequestration

4.3.3.1. 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 removes carbon from the atmosphere (Tiefenbacher et al., 2021). In LCA, this represents avoided climate change impacts, where farms using compost should receive environmental credits for sequestering CO_2. 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). 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).

4.3.3.2. 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., 2023). Some compost LCAs (from a biowaste treatment perspective) have used carbon sequestration at rates of 10–14% of organic carbon (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 the farm's GHGs. In a different UA LCA, Dorr et al. (2023a), applied standard carbon sequestration rates using sensitivity analysis to estimate emissions offsets between 3% and 23%. 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 (Nicese et al., 2021; Strohbach et al., 2012).

4.3.3.3. Recommendations. Excluding carbon sequestration from compost (or other organic inputs) in the main results of UA LCAs is recommended 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.

4.3.4. Compost emission factors

4.3.4.1. Challenge. The most impactful component of the compost life cycle is 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 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).

4.3.4.2. 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. The LCA database Ecoinvent (Wernet et al., 2016) 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 climate change impacts were reduced by 2–14% when using the inventory from Ecoinvent rather than from a meta-analysis by Nordahl et al. (2023). For more complete summaries of measured composting emission factors, see review papers by Nordahl et al. (2023), Boldrin et al. (2010), and Amlinger (2008), and discussion section reviews in Quirós et al. (2015) and Avadí et al. (2020).

4.3.4.3. Recommendations. Scenarios involving different emission factors should be modeled 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.

4.4. Substrate

4.4.1. 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 structural limitations since rooftops cannot always support heavy soil loads. In these cases, soilless cultivation methods (such as hydroponics, aeroponics, or aquaponics) or imported substrates are utilized. 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). Several UA LCAs have found that creating and replenishing substrate was the largest contributor in most LCA indicators (Kim et al., 2018; Vacek et al., 2017).

Substrate's lifetime directly affects its impacts, but very little information is available on this parameter, which may be as low as three years for perlite (Rufí-Salís et al., 2020a). 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, such lifetimes may be shorter than anticipated (Demailly and Darly, 2017).

4.4.2. 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 (Dorr et al., 2021b), spent brewer's grain, and shredded paper have also been observed (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

Table 3
Greenhouse gas (GHG) emissions per ton of fresh waste composted from some of the main sources of composting used in urban agriculture. Emissions of N_2O and CH_4 are shown in kilograms of substance per ton of compost. OFMSW: organic fraction of municipal solid waste. Literature sources: a) Andersen et al., 2011, b) Martínez-Blanco et al. (2010), c) Colón et al. (2010), d) Quirós et al., 2015, e) Wernet et al. (2016), f) Asselin-Balençon et al., 2020, g) Nordahl et al. (2023).

Reference	Type of composting system	N_2O emissions	CH_4 emissions	GHG emissions	Notes (CO, NH_3, VOC emissions)
		(kg/ton fresh waste)			
Andersen et al., 2011[a]	Home composting, closed unit	0.30–0.55	0.4–4.2	100–239	6 composting units
Martínez-Blanco et al., 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 et al., 2010c	Fruit and vegetable scraps, yard waste, home composting	0.2	0.3	67.1	VOCs = 0.32, NH_3 = 0.03.
Quirós et al., 2015 HE[d]	Home composting, high-emission system	1.16	1.35	379.4	Leftover fruits and veg, yard waste. NH_3 = 1.3.
Quirós et al., 2015 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 et al., 2023 YW[g]	Yard waste, average from review	0.043	2.31	70.6	Average of 9 values
Nordahl et al., 2023 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 et al., 2023 manure[g]	Manure, average, from review	0.354	2.82	176.0	Average of 41 and 45 values for CH_4 and N_2O

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.

4.4.3. Recommendations

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. Residual waste products that have negligible economic value should not incur impacts from the first use, according to economic allocation principles (ISO, 2006). For both valuable byproducts and residual waste products, impacts from their transportation 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 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. Results can be sensitive to this assumption, thus 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 to 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 4.3.4, and impacts of composting should be allocated between the waste generator and the compost user. If substrate is applied as a soil amendment 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 along with physio-chemical characteristics (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.

4.5. Transportation and delivery

4.5.1. 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—let 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 transportation (often 10–25%, but as high as 54%), due to the potential relatively lower impacts at the farm stage, long distances, refrigerated transportation, and airplane travel (Barbier et al., 2019; Bell and Horvath, 2020). The benefit of reduced transportation is mostly tested through comparisons of UA LCA results to the supply chains of rural agriculture (Dorr et al., 2021a). Challenges arise here in defining consistent system boundaries between urban and rural agriculture.

Post-farm transportation in UA is often directly to the consumer. This is especially evident when products from UA are delivered by walking or by bike because there are almost no impacts. In these cases, the system boundary *implicitly includes* the nil transportation to the consumer (Sanyé-Mengual et al., 2018b; 2013). The final step of transportation 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 Fig. 3) (Pérez-Neira and Grollmus-Venegas, 2018). It is difficult to model consumer transportation behavior and to isolate transportation specifically for food purchases from other transportation. Therefore, many comparisons between urban and rural agriculture products risk comparing a cradle-to-consumer UA system with a cradle-to-market rural agriculture system, although it may involve large transportation distances over long supply chains (Majewski et al., 2020). 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 transportation in UA. Differences in packaging, which can vary between conventional and alternative retail options packaging, may also enhance the benefits of UA. GHG savings larger than 50% were reported for tomatoes, for example (Sanyé-Mengual et al., 2013).

4.5.2. Examples

Transportation 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 (Fig. 3) (Sanjuan-Delmás et al., 2018; Torres Pineda et al., 2020). Several UA LCAs include distribution by car to the consumer, based on a simplified model/distribution of transportation modes and distances from the distribution point to consumers' homes (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 transportation impacts (Melkonyan et al., 2020; Stelwagen et al., 2021).

4.5.3. 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 transportation, displaying 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. The delivery scheme of a case study should be clearly described, including the transportation distances, modes, and frequencies of deliveries. Careful consideration must be taken to ensure that system boundaries are consistent. A cradle-to-consumer boundary is implied and should be considered for comprehensiveness and to account for the related environmental benefit of UA. Consumer transportation should be included for the rural system as it is not represented in LCA databases (Fig. 3).

4.6. Variability and uncertainty of UA

4.6.1. Challenge

Agricultural LCAs have particular issues with high variability because of diversity in controlled factors like farming practices and logistics and in environmental factors like climate and soil characteristics (McLaren, 2010; Notarnicola et al., 2017). Controlled factors are likely more variable in UA than in rural agriculture. Urban settings introduce physical limitations (e.g., shading from buildings, poor-quality anthropogenic soils, air pollution, and limited access to materials) which spur innovative growing practices and setups (Taylor, 2020; Wagstaff and

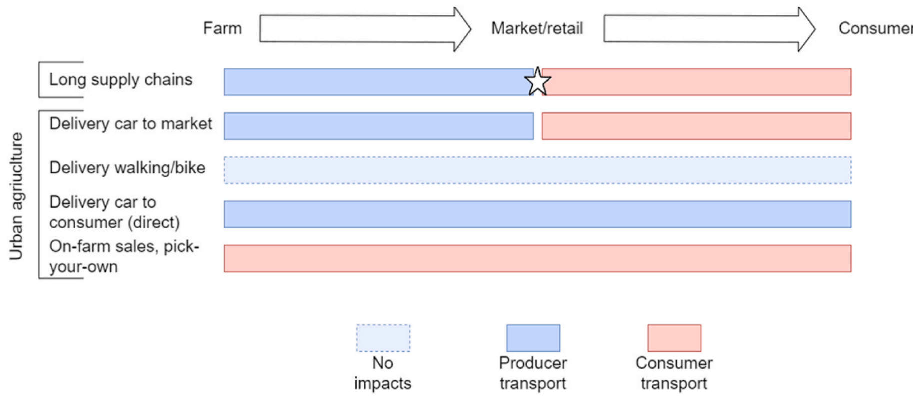

Fig. 3. Modelling of consumer transportation for urban agriculture produce. The downstream system boundary is shown for several simplified distribution schemes. Colored bars indicate who is doing the travel, and the empty bar for walking/bike indicates that there are no environmental impacts from this travel. Many rural food LCAs have a system boundary that ends at the market/retail stage (shown by the star).

Wortman, 2015). Human elements such as motivation for urban farming, farmer 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.

4.6.2. *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). 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., 2023a).

4.6.3. *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 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). Variability 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. More complete, transparent descriptions of case studies would help readers interpret results and make more relevant, accurate comparisons between studies.

5. Research directions for UA LCAs

This section presents aspects of UA LCAs that require future research. These topics should not necessarily be systematically included in UA LCAs because more research and development are needed. Nonetheless, this section presents practical recommendations for including them in UA LCAs now. It also discusses research directions that can improve UA LCAs, and how applying LCA to UA can lead to insights for LCA overall.

5.1. *Align with urban land uses and green infrastructure LCAs*

5.1.1. *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 and 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). UA is one option for urban green infrastructure amongst many, and may be more comparable to a park or other social/recreational activity than 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 have not emerged in UA LCAs.

5.1.2. *Recommendations*

Increased attention should be paid to 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. Alternatively, the breakdown of revenue from food sales compared to other sources of funding could be used for economic allocation.

5.2. UA in the context of urban symbioses

5.2.1. 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 ripple effects on municipal organic waste treatment, wastewater treatment or urban transport logistics (Sanjuan-Delmás et al., 2018). Thus, in the wider sense, UA contributes to "urban symbiosis" (Yang and Yang, 2022), generating positive (or negative) impacts at this level. Assessing these effects may be considered a consequential form of LCA, and 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, but requires estimates of the current diets of city inhabitants (to evaluate substitution effects) (Dorr et al., 2022), 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). 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).

5.2.2. Recommendations

Researchers should apply LCA to UA at the city scale, which highlights multiple symbioses and can account for context-specific aspects. Farm-level LCAs should include descriptions of the city to facilitate interpretation, such as the position of the farm in relation to the city center/boundary, city density, and the role of UA in the city (e.g., in its history or social movements). Location and climatic characteristics are crucial for estimating potential benefits of UA implementation.

5.3. Ecosystem services and positive impacts

5.3.1. Presentation

LCA is designed to evaluate the negative (adverse) impacts of a system rather than its positive impacts (benefits). The ecosystem services (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; Tang et al., 2018), 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 (e.g., 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 an FU based on food production, they essentially quantify the impact of ES. Boone et al. (2019) demonstrated a method to allocate between 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 4.3.3. Reduction of the urban heat island effect is a frequently proposed regulation of 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., 2018). 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 in section 5.4).

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., 2018) 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, usually driven by over-exploitation of resources and habitat alteration in relation to land use (Crenna et al., 2020). LCA models the upstream and downstream impacts of materials and processes on biodiversity around the world, and considering local biodiversity remains a challenge (Crenna et al., 2020). 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).

5.3.2. 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.

5.4. Social benefits and life cycle sustainability assessment

5.4.1. Presentation

A main strength of UA is its multifunctionality, with important social functions and multiple impacts and potential benefits (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, it may be relevant to use a functional unit 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).

The LCA community has promoted life cycle sustainability assessment, which combines environmental LCA, life cycle cost analysis (see Peña and Rovira-Val (2020) for its application to UA), and S-LCA. Such holistic life cycle sustainability assessments are still largely aspirational (Fauzi et al., 2019; Finkbeiner et al., 2010). Practitioners are urged 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).

5.4.2. 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 FUs to account for social aspects of UA. Although researchers 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.

6. Conclusions

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 in defining what questions to be addressed with LCA. In this article, recommendations and research directions were we laid out that are intended to improve LCAs of UA. These improvements can lead to more thorough LCAs and more consistency between case studies. The questions that UA LCAs may aim to answer were also outlined, in 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 policymakers 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.

CRediT authorship contribution statement

Erica Dorr: Conceptualization, Investigation, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. **Benjamin Goldstein:** Conceptualization, Methodology, Supervision, Writing – review & editing. **Christine Aubry:** Conceptualization, Funding acquisition, Methodology, Project administration, Supervision, Writing – review & editing. **Benoit Gabrielle:** Conceptualization, Funding acquisition, Methodology, Project administration, Supervision, Writing – review & editing. **Arpad Horvath:** Conceptualization, Methodology, Project administration, Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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Appendix A. Supplementary data

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