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

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1 How to do life cycle assessments of urban agriculture

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11 Highlights

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

17

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20 Abstract

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

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

37

38 **1 Introduction**

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

55 General LCA frameworks and guides have been proposed to improve the rigor and
56 comparability of LCAs, and include the International Organization for Standardization (ISO)
57 framework (ISO, 2006a, 2006b), the ILCD handbook (European Commission, 2010a), and the
58 Product Environmental Footprint Category Rules Guidance (European Commission, 2017).
59 For LCA of specific sectors, methodological guidelines have highlighted unique aspects that
60 require special attention. Failure to account for these aspects can skew results and hamper
61 decision making. For instance, the inclusion of direct and indirect land use change in biofuel
62 production fundamentally altered the carbon calculus of this technology, and caused a
63 reappraisal of government policies to support first-generation biofuels (Searchinger et al.,
64 2008).

65 To avoid similar mistakes in other fields, researchers have produced LCA guidelines for
66 diverse industries and technologies ranging from waste management (Laurent et al., 2014) to
67 bioplastics (Bishop et al., 2021). In the area of food, best practices exist for LCAs of crop
68 production (Adewale et al., 2018), organic agriculture (van der Werf et al., 2020), fruit
69 orchards (Cerutti et al., 2014), vegetables (Perrin et al., 2014), climate-smart agriculture
70 (Acosta-Alba et al., 2019) and agricultural use of microbial inoculants (Kløverpris et al.,
71 2020). Other work has evaluated the combination of agricultural LCAs with circular economy
72 (Stillitano et al., 2021) or ecosystem service assessments (Tang et al., 2018). Not to mention
73 the large body of work reviewing the methodological choices, challenges, and best practices
74 of agricultural LCAs in general (Audsley et al., 1997; Brentrup et al., 2004; Caffrey and Veal,
75 2013; Cucurachi et al., 2019; Dijkman et al., 2018; McLaren, 2010; Nemecek and Gaillard,
76 2010; Notarnicola et al., 2017, 2012). Such methodological reflections and recommendations
77 have not yet been done for UA.

78 This study intends to fill this gap by providing a guideline for how to assess UA using LCA.
79 It is applicable to all forms of UA in its most general definition of “food production in and
80 around cities” (Mougeot, 2000). It builds on observations from a previous literature review
81 and meta-analysis of the environmental impacts of UA (Dorr et al., 2021a) to provide
82 practical recommendations when applying LCA to UA, and takes a more comprehensive
83 approach to both UA and LCA. This guideline was also tested and iteratively refined during a
84 recent LCA of a diverse set of urban farms in France and the United States (Dorr et al.,
85 2022b).

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

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

97 **2 Why do LCAs of urban agriculture?**

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

112 Table 1 The goal of a life cycle assessment should answer a larger question. Some relevant questions
 113 for life cycle assessment of urban agriculture are presented here, along with a description/justification
 114 for each question and possible functional units (FU). Some questions are already prevalent in the
 115 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

<p>infrastructure to implement in a city? How does UA affect the GHG emission or other environmental impacts of a city?</p>	<p>City leaders must decide which types to implement. 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.</p>	<p>Urban metabolism, land area, operation of other sectors (i.e. waste treatment)</p>
<p>What are potential trade-offs of socially motivated UA projects?</p>	<p>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?</p>	<p>Single crop, mixed crops, land area, cost/revenue, total operations of urban farm, social functions (i.e. hours of education, number of participants)</p>
<p>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...)?</p>	<p>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.</p>	<p>Single crop, mixed crops, land area, cost/revenue, total operations of urban farm, social functions (i.e. hours of education, number of participants)</p>
<p>How can UA be designed or managed to minimize environmental impacts?</p>	<p>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.</p>	<p>Any</p>

116

117 **3 Challenges and practical recommendations for UA LCAs**

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

126 **3.1 Crop diversity**

127 *Challenge:*

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

143 *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: <ul style="list-style-type: none"> • 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.

144 Most UA LCAs with high crop diversity chose FUs covering total annual operations of a farm
145 or impacts per unit area (Martinez et al., 2018; Pérez-Neira and Grollmus-Venegas, 2018;
146 Sanyé-Mengual et al., 2018a). This avoids highly uncertain allocations, considers additional
147 functions of agriculture, and facilitates cross-farm comparisons. However, results are difficult
148 to extrapolate since they represent production of varied crops which are usually not
149 functionally comparable, and sometimes the crops grown are not communicated. Another
150 strategy uses published data or farmer estimates to estimate an life cycle inventory for each

151 crop (Caputo et al., 2020; Kulak et al., 2013; Liang et al., 2019). This allows for crop-level
152 analysis for polycultures, but accuracy is inevitably lost when equating UA to other systems.
153 For instance, when these data come from rural agriculture, representativeness of UA is likely
154 sacrificed. Other researchers have allocated between many crops based on mass, area, calorie
155 content, nutritional index, or time of cultivation of each crop, to generate results per crop
156 (Pennisi et al., 2019; Rufí-Salís et al., 2020a; Sanyé-Mengual et al., 2015b). Finally, some
157 researchers used a simplified FU covering a basket of crops (i.e. 1 kilogram of mixed lettuce,
158 tomato, and pepper) (Boneta et al., 2019; Hu et al., 2019). These results are difficult to use
159 elsewhere since unique mixes of crops are not precisely comparable, and authors may not
160 include which crops are included in the mix or in what proportions. LCAs of rural farms with
161 many crops have also used a FU of kilogram of mixed crop (Christensen et al., 2018; Pepin,
162 2022), which complicates interpretation.

163 ***Recommendations:***

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

175 **3.2 Data availability**

176 ***Challenge:***

177 Data collection in LCA is often highly labor-intensive. For an agricultural LCA, data on farm
178 inputs and outputs are needed. In conventional agriculture, primary data come from farmer
179 interviews, receipts, or informed estimates/calculations (Christensen et al., 2018). Secondary
180 data, such as the UC Davis Crop Budgets (Caffrey and Veal, 2013), can address data gaps or
181 create entire inventories. Similar quality data are rare for UA because urban farmers usually
182 keep limited records (Cleveland, 1997; Egerer et al., 2018; Whittinghill and Sarr, 2021).
183 Inputs and food production in UA (especially informal UA) can be extremely variable and
184 difficult to predict, casting doubt on the applicability of secondary data for UA (Dorr et al.,
185 2022c). Collective and community-based UA may have many participants who harvest and
186 use agricultural inputs, which further complicates record keeping. Self-reporting and
187 participatory methods face issues of reliable, consistent data collection and participant fatigue
188 (CoDyre et al., 2015).

189 ***Examples:***

190 The available UA LCAs are based on both primary and secondary data. Data for UA LCAs
191 come from many different sources, including directly measured data, operations records,
192 farmer interviews and surveys, and secondary data from urban or rural agriculture (Dorr et al.,
193 2021a). Data sources and data collection difficulties are largely discussed in research on UA
194 practices in general, but not so much in UA LCAs (McDougall et al., 2019; Pollard et al.,
195 2018).

196 ***Recommendations:***

197 Due to the variability and lack of data regarding UA practices, collecting primary data from
198 case studies should be prioritized. Past records of operation may be used, although it is
199 unlikely that urban farmers have records of all necessary information for an LCA. A data
200 collection campaign, with commitment from farmers, may be necessary. Researchers should
201 discuss data needs with farmers early and often to identify the most feasible methods to
202 collect data, create a data collection plan, and regularly follow up to ensure reliability. This is
203 a crucial step because unclear or overly burdensome data collection efforts may be abandoned

204 or unusable. Researchers should consider the types of data that may already be collected at
205 urban farms (i.e. level of detail, time frame, units), and adapt the data collection plan
206 accordingly. Surveys, growing logs, and harvest notebooks should be co-designed with
207 farmers to track harvest and inputs (Nicholls et al., 2020). Water use should be measured
208 using water meters or calculated using the number of buckets or watering cans used and their
209 volume (Pollard et al., 2018). Researchers should periodically check for leaks in irrigation
210 systems, which may be substantial (Dorr et al., 2022c). Soil amendments, such as compost
211 and fertilizers, should be tracked through the amount applied, or the amount
212 purchased/delivered (although this may require temporal allocation to growing season). The
213 detailed description of our data collection methods with UA case studies in the appendix of
214 (Dorr et al., 2022b) provides concrete examples of how to collect data across diverse systems.

215 **3.3 Compost**

216 Compost is the main input to many urban farms (see detailed review in the Appendix B)
217 (Cofie et al., 2006; Dobson et al., 2021; Edmondson et al., 2014). A proposed environmental
218 advantage of UA is its potential to grow food and reduce landfill burdens by applying
219 compost from urban organic waste (Mohareb et al., 2017; Specht et al., 2014). Compost is
220 thus central to UA despite infrequent and inconsistent quantification in UA LCAs (Dorr et al.,
221 2021a). Even for rural agriculture LCAs, compost is often omitted, or its inclusion is
222 inconsistent and unclear (Bartzas et al., 2015). Surprisingly, compost is not explicitly
223 mentioned in reviews of LCAs of organic agriculture, where it is expected to be extensively
224 used (Meier et al., 2015; van der Werf et al., 2020). LCAs focusing on compost use in
225 agriculture found that the GHGs emitted from microbial decomposition (CH_4 and N_2O) are a
226 major contributor to climate change impacts, and avoided burdens (i.e. subtracting emissions
227 from avoided processes, such as avoided incineration of organic waste) and allocation have
228 large effects on the results for rural agriculture (Bartzas et al., 2015; Christensen et al., 2018;
229 Martínez-Blanco et al., 2009) and for UA (Dorr et al., 2022b, 2017; Liang et al., 2019; Martin
230 et al., 2019). Therefore, compost is given extra attention for this section.

231 3.3.1 Off-farm compost system modeling

232 **Challenge:**

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

240 **Examples:**

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

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

262 ***Recommendations:***

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

270 **3.3.2 On-farm compost system modeling**

271 ***Challenge:***

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

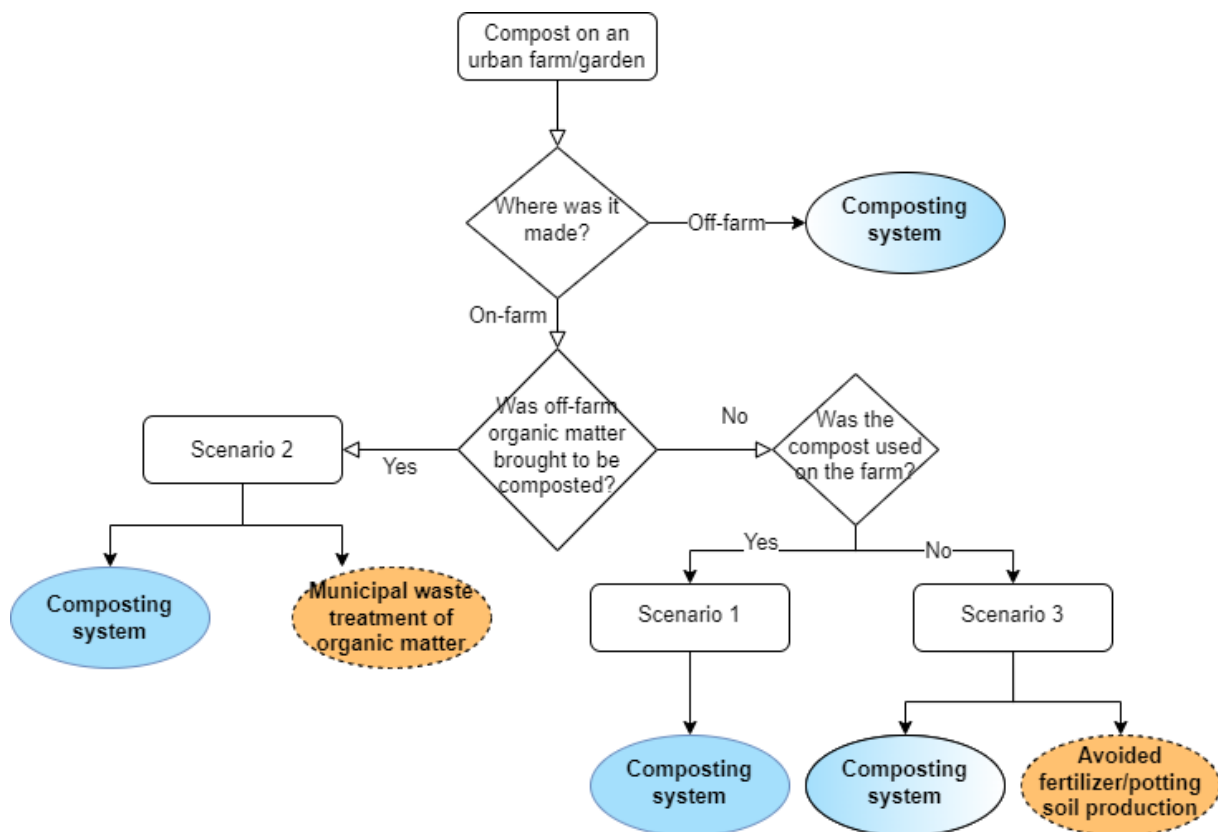


Figure 1 A decision tree clarifies the different scenarios of composting for an urban farm and how to account for composting impacts. Blue circles represent impacts from composting emissions, and orange circles with dotted outlines represent substituted processes that can be subtracted from the farm’s 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.

- 275 Scenario 1) made using on-farm biomass and used on the farm,
- 276 Scenario 2) made using on-farm biomass plus other green waste brought to the farm, and used
- 277 on the farm, or
- 278 Scenario 3) made using on-farm biomass and not used on the farm (i.e., for hydroponics
- 279 systems that generate biomass waste but do not use compost).

280 These possible scenarios, and the relevant system modeling decisions for LCA, have not been

281 explicitly examined before.

282 ***Examples and recommendations:***

283 Scenario 1 is a type of “closed loop” recycling system where the waste is generated and the

284 recycled product is used within the same system. Examples of this are in Boneta et al. (2019)

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

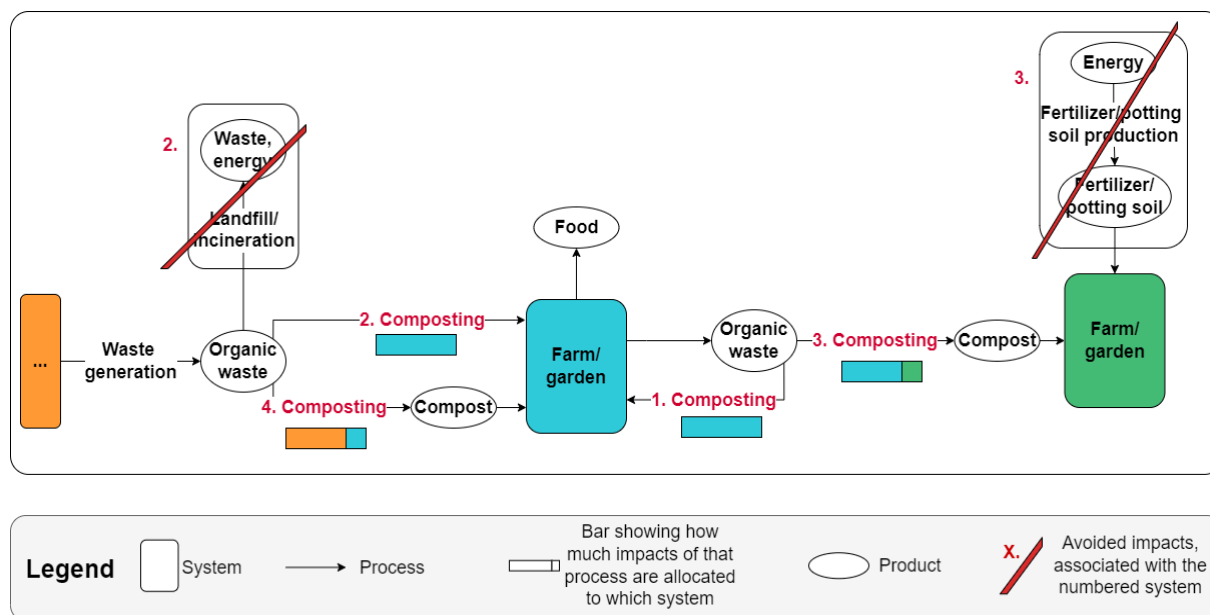


Figure 2 A process diagram shows the different composting scenarios as described in the text, from the perspective of the blue farm in the center. The numbers refer to the scenarios described in section 3.3.2, and scenario 4 refers to off-farm compost, described in section 3.3.1.

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

299 accounted for in the LCA results by showing smaller impacts than if the farm had used
300 fertilizer.

301 Scenario 3 composting can be found at urban farms that create inedible biomass waste (all
302 farms) but do not use compost, such as soilless hydroponics or aeroponics systems. This type
303 of composting represents a multifunctional process: it treats the farm's waste and creates a
304 compost product. Here the UA site is a waste generator, as discussed in the off-farm compost
305 section (section 3.3.1). Farms should be credited with avoided environmental burdens from
306 production of the fertilizer or potting soil that the produced compost substitutes (Corcelli et
307 al., 2019; Goldstein et al., 2016). Vieira and Matheus (2019) provide a comprehensive review
308 and recommendations on the matter. Composting for waste treatment of biomass can account
309 for 10-15% of climate change impacts in UA (Corcelli et al., 2019; Sanjuan-Delmás et al.,
310 2018), but avoided burdens of fertilizer production can generate net impact abatements
311 (Corcelli et al., 2019).

312 3.3.3 Carbon sequestration

313 *Challenge:*

314 Compost is rich in organic carbon that is stabilized and stored after application to soil (Lal et
315 al., 2015). Carbon sequestration through composting with low-carbon soil management can
316 remove carbon from the atmosphere (Tiefenbacher et al., 2021). From an LCA perspective,
317 this represents avoided climate change impacts, where farms using compost should receive
318 environmental credits for sequestering CO₂ as organic carbon in their soils. However, the
319 long-term fate of organic carbon is mostly unknown and highly context dependent because of
320 complex soil ecology. This introduces high uncertainty to a process that can largely influence
321 LCA results (McClaren, 2010; Strohbach et al., 2012; Tidåker et al., 2017). Existing agriculture
322 soil carbon models are highly time and data intensive, and are poorly adapted to UA where
323 unique substrate and high composting rates predominate (Dorr et al., 2017). .

324 *Examples:*

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

339 ***Recommendations:***

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

344 **3.3.4 Compost emission factors**

345 ***Challenge:***

346 The most impactful component of the compost life cycle is gaseous emissions of methane,
347 nitrous oxide, ammonia, and volatile organic compounds during the composting process
348 (Boldrin et al., 2009; Pergola et al., 2020). These emissions strongly affect climate change,
349 acidification, eutrophication, and photochemical ozone formation impacts (Pergola et al.,
350 2020). High variability in gaseous emissions from composting—due to differences in
351 technical systems, feedstocks, and composting practices—result in high variability in

352 composting impacts (Joint Research Centre, Institute for Environment and Sustainability,
353 2012).

354

Reference	Type of composting system	N ₂ O	CH ₄	GHG	Notes (CO, NH ₃ , VOC emissions)
		emissions	emissions	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	Retrieved 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 ₂ O
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 ₂ O

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-Balencçon et al., 2020, g) Nordahl et al., 2022.

356 ***Examples:***

357 Many UA LCAs use composting emission factors from Andersen et al. (2012, 2011),
358 Martínez-Blanco et al. (2010), and Colón et al. (2010), because they measured inventory data
359 specifically for home composting, which is representative of small scale, on-farm composting
360 operations. The LCA database Ecoinvent (Wernet et al., 2016), which uses inventory data
361 from Edelmann and Schleiss (1999), is also commonly used to model composting in UA and
362 conventional agricultural LCAs. Table 3 shows the wide range in composting GHG emission
363 factors from sources commonly used in agricultural LCAs. This selective list of emission
364 values highlights the potential pitfalls from selecting composting inventories with such
365 variability. Indeed, in our case study we found that climate change impacts were reduced by
366 2-14% when we used the inventory from Ecoinvent rather than from a meta-analysis by
367 Nordahl et al. (2022). For more complete summaries of measured composting emission
368 factors, see reviews papers by Nordahl (2022), Boldrin (2010), and Amlinger (2008), and
369 discussion section reviews in Quiros (2015) and Avadi (2020).

370 ***Recommendations:***

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

378 **3.4 Substrate**

379 ***Challenge:***

380 A unique characteristic of UA compared to rural agriculture is that it is not necessarily carried
381 out on soil. Soil, or top-soil, is defined as natural bodies made of organic and inorganic
382 material that are formed at the surface as the result of complex biogeochemical and physical

383 processes (Brevik and Arnold, 2015; Hartemink, 2016). Using soil as a growing medium is
384 often not an option in UA due to soil pollution in cities, or lack of greenfields. In these cases,
385 soilless cultivation methods are used (such as hydroponics, aeroponics, or aquaponics), or a
386 substrate/growing medium is imported. In an LCA, substrate can be considered infrastructure
387 that requires material inputs of large quantities and variable types. Current practices around
388 producing substrate in UA LCAs are unclear, because it often goes unmentioned, it seems to
389 be inconsistently included, and system modeling decisions around the recycled materials often
390 incorporated in substrate are variable (Dorr et al., 2021a). Yet, several UA LCAs have found
391 that creating and replenishing substrate was the largest contributor for most impact categories
392 (Dorr et al., 2017; Kim et al., 2018; Vacek et al., 2017).

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

401 ***Examples:***

402 Peat, coir, wood and compost are commonly used to produce substrate (Barrett et al., 2016).
403 In UA, materials such as crushed brick, spent coffee grounds, spent brewer's grain, and
404 shredded paper have also been observed (Dorr et al., 2021b; Grard et al., 2020; Martin et al.,
405 2019). The numerous possible substrate inputs, mostly co- or by-products, lead to many
406 options for modeling the materials.

407 Limited details are available regarding lifetime and fate of permanent substrates in UA LCAs.
408 Dorr et al. (2017) evaluated a research-oriented rooftop farm using substrate in raised beds,
409 and assumed a 10-year farm lifetime and that substrate had no end-of-life treatment (as it

410 would be donated and reused). Kim et al. (2018) evaluated a rooftop farm and green roof, and
411 assumed a 40-year lifetime based on the durability of the roof membrane material. Vacek et
412 al. (2017) did an LCA of green roofs and assumed a lifetime of 20 years, noting that they
413 would require renovation after this point. They assumed that substrate would be landfilled,
414 being too degraded for recycling/reuse.

415 ***Recommendations:***

416 Regarding materials, the LCA guidelines published by Growing Media Europe (2021) detail
417 how to model and what to include for numerous substrates found in UA. Peat and peat moss
418 have been well studied, and the processes available in LCA databases should be used. Impacts
419 for coconut and wood/bark-based materials should be allocated on an economic basis between
420 the main coconut and forestry products and the substrate byproducts (European Commission,
421 2010b). Residual waste products that have negligible economic value should not incur
422 impacts from the first use, according to economic allocation principles (ISO, 2006b). For both
423 valuable byproducts and residual waste products, impacts from their transport after the
424 original site of use, and energy and water needed for processing into substrate should be
425 accounted for (Growing Media Europe, 2021).

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

434 Replenishing substrate helps maintain substrate volume and quality. Impacts of these
435 replenishments should be temporally allocated to the time between applications. For example,

436 if substrate is replenished every two years, then half of the amount applied can be allocated
437 the system in an LCA considering one year of production.

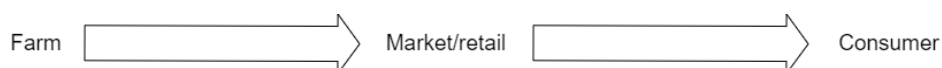
438 End-of-life for inorganic substrates will likely include municipal waste treatment or recycling.
439 Organic substrates are mostly composted or applied to fields as a soil improver (Growing
440 Media Europe, 2021). For composting, the farm can be seen as the waste-generator described
441 in section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

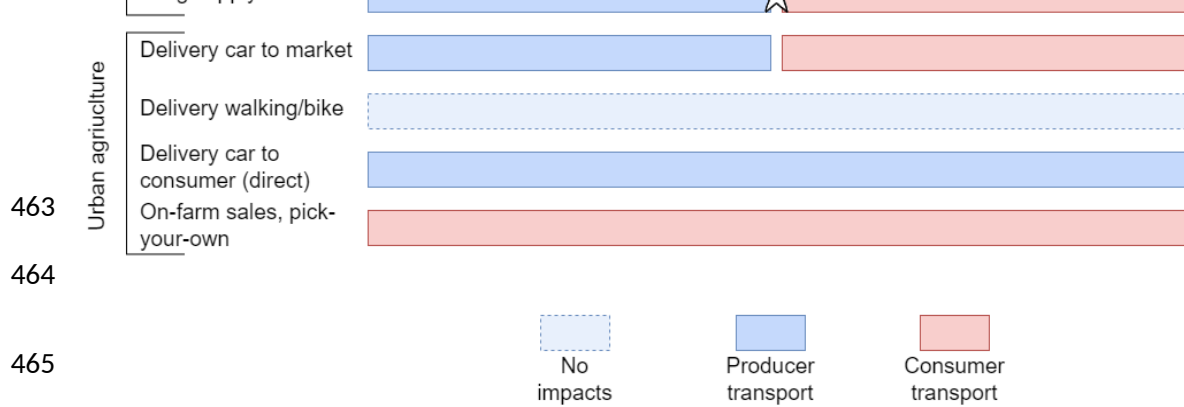
445 Increased transparency and improved reporting regarding substrates are necessary in UA
446 LCAs. The nature and the origins of substrate material should be clearly described, plus
447 physio-chemical characteristics, if available (Barrett et al., 2016). The amount of substrate
448 initially applied,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

450 **3.5 Transport and delivery**

451 *Challenge:*

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





463 products from ITA are delivered by walking or by bike because there are almost no impacts
 464 Figure 3 The downstream system boundary is shown for several simplified distribution schemes.
 465 (Colored bars indicate who is doing the travel, and the empty bar for walking/bike indicates that there
 466 are no environmental impacts from this travel. Many rural food LCAs have a system boundary that
 467 ends at the market/retail stage (shown by the star). Several types of distribution networks that are
 468 common for UA are inconsistent with this system boundary, because there is no equivalent
 469 market/retail stage. UA LCA practitioners should ensure consistent system boundaries when
 470 comparing results with rural food LCAs.
 471 to/by the consumer, also called the ‘last mile’, is usually not included in food LCAs, and the
 472 system boundary ends at the market/retail stage (the star in Figure 3) (Pérez-Neira and
 473 Grollmus-Venegas, 2018). It is usually not included because it is difficult to model consumer
 474 transport behavior, and to isolate transport specifically for food purchases from other
 475 transport. Therefore, many comparisons between urban and rural agriculture products risk
 476 comparing a cradle-to-consumer UA system with cradle-to-market rural agriculture system.
 477 The last mile step is important because it can account for an even larger food transport
 478 distance (or ‘food miles’) than the transport over long food supply chains (Majewski et al.,
 479 2020). Impacts of the last mile for food (from customer travel) can contribute up to 21% of
 480 life cycle climate change impacts of pasta (Gnielka and Menzel, 2021), or 6% of urban food
 481 system climate change impacts (Stelwagen et al., 2021). Therefore, inaccurate comparisons
 482 here may omit a large benefit of reduced consumer transport of UA.

482 **Examples:**

483 Transport from the farm to the consumer on foot or by bike, or when production occurs in or
 484 on a building where consumers live or work, has been considered in several UA LCAs. They
 485 state that there are no processes or impacts for delivery (Figure 3) (Sanjuan-Delmás et al.,
 486 2018; Sanyé-Mengual et al., 2018a; Torres Pineda et al., 2020). Several UA LCAs include
 487 distribution by car to the consumer, based on a simplified model/distribution of transport
 488 modes and distances from the distribution point to consumers’ homes (Hall et al., 2014; Hu et
 489 al., 2019; Pérez-Neira and Grollmus-Venegas, 2018). Other LCAs regarding urban food

490 consumption and food products have focused on the last-mile transport impacts (Bevilacqua
491 et al., 2007; Melkonyan et al., 2020; Stelwagen et al., 2021).

492 ***Recommendations:***

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

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

510 Due to the difficulty of modeling these complex distribution networks, in-depth research on
511 this topic may need to be done separately from production-focused UA LCAs (Coley et al.,
512 2009; Stelwagen et al., 2021). This represents an opportunity for cross-disciplinary research
513 on UA production and urban mobility. A city or foodshed scale may provide additional
514 insight, as this topic quickly veers into the larger urban food logistics system rather than farm
515 systems (Benis and Ferrão, 2017; Melkonyan et al., 2020).

516 3.6 Variability and uncertainty of UA

517 *Challenge:*

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

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

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

543 understanding the general performance of UA, since there is not really a ‘general’ situation
544 for UA.

545 ***Examples:***

546 One of the most common ways of addressing variability and uncertainty in UA LCAs is
547 presenting alternative scenarios in the form of sensitivity analyses. This was done in UA
548 LCAs by modeling different infrastructure lifetimes (Dorr et al., 2017; Martin and Molin,
549 2019), crop yields (Romeo et al., 2018; Rufí-Salís et al., 2020b), or light efficiency for indoor
550 systems (Pennisi et al., 2019; Shiina et al., 2011). Another strategy was to use ranges of
551 inventory values, for example for delivery/distribution schemes, generating a range of results
552 (Hu et al., 2019; Pérez-Neira and Grollmus-Venegas, 2018; Stelwagen et al., 2021). When
553 parameters with high variability are identified, the goal of the LCA can shift to find tipping
554 points where one system performs better/worse than another (usually UA vs rural). This was
555 done for yield and distance from producer to consumer (Kulak et al., 2013; Sanyé-Mengual et
556 al., 2015a). Alternatively, Monte Carlo simulations can be employed to quantify ranges of
557 results based on distribution of parameters, such as composting emission factors and bulk
558 density (Dorr et al., 2022b).

559 ***Recommendations:***

560 Variability and uncertainty within systems can be reduced or accounted for with several
561 strategies. Temporal variability, due to annual climate differences or changes in operations
562 (for example due to farmer turnover), should be reduced by collecting data for multiple years
563 and using an average of values, or selecting the most representative year (Loiseau et al.,
564 2020). Specialized indicators can be used that quantify how important variability is for a
565 system (Hauck et al., 2014). Variability of inventory items should be considered using
566 distributions or ranges (Stelwagen et al., 2021), and probabilistic simulations, such as Monte
567 Carlo simulations (Huijbregts, 1998).

568 Variability between systems is problematic when trying to compare or summarize results for
569 similar systems. Such comparisons are necessary to draw trends and generalize LCA findings,

570 which is a feature of rather mature LCA research topics. Few technical recommendations can
571 be made here, but we note that more complete, transparent descriptions of case studies would
572 help readers interpret results and make more relevant, accurate comparisons between studies.

573 **4 Research directions for UA LCAs**

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

579 **4.1 Align with urban land uses and green infrastructure LCAs**

580 ***Presentation:***

581 The UA LCA literature is dominated by a product-based perspective, which inherently places
582 the focus on the food-production function of UA. UA distinguishes both the unique, non-rural
583 position of agriculture, plus the non-conventional use of urban space (Neilson and Rickards,
584 2017). The latter perspective has not been widely studied with LCA, except for studies
585 comparing different uses of rooftops for flower gardening, farming, or solar panels (Corcelli
586 et al., 2019; Goldstein et al., 2016; Kim et al., 2018). UA is one option for urban green
587 infrastructure among many others, and may be more comparable to a park or other
588 social/recreational activity than it is to rural agriculture. There is a wealth of literature on
589 environmental assessments of urban parks and forests (Strohbach et al., 2012), golf courses
590 (Tidåker et al., 2017), urban wetlands (Duan et al., 2011), grassy areas (Smetana and
591 Crittenden, 2014), and other green infrastructure (Nicese et al., 2021), and it would be useful
592 to relate UA to these land uses. It could provide meaningful comparisons to similar systems,
593 and illuminate shortcomings in UA LCAs that are obscured by a product-based perspective.
594 For example, urban green infrastructure LCAs found that waste treatment of biomass can be
595 highly impactful (Nicese et al., 2021; Tidåker et al., 2017), and results can be highly sensitive

596 to carbon sequestration (Strohbach et al., 2012; Tidåker et al., 2017), which has not emerged
597 in UA LCAs.

598 ***Recommendations:***

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

606 **4.2 City-scale/Scaling up**

607 ***Presentation:***

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

613 The effects on the city of “scaling up” or developing UA has been modeled under different
614 scenarios. Goldstein et al. (2017) evaluated the effect of installing UA in available land in
615 Boston, USA, and found that it could reduce food-related climate change impacts at the city
616 level by 1-3%, and increase land occupation by 1%. Mohareb et al. (2018) performed a
617 similar analysis for the USA and found food sector GHG emissions decreased by 1%. Other
618 scaling-up analyses suggest that UA could ‘absorb’ and compost 9% of municipal organic
619 waste in Boston (Goldstein et al., 2017), 17% in Lyon, France, and 52% in Glasgow, Scotland
620 (Weidner and Yang, 2020). Extrapolating farm-level results to the city-scale helps provide
621 perspective, because if for example fruits and vegetables are substantially more or less
622 impactful than rural products, but at the city or individual diet scale the difference is meagre,

623 then UA LCAs should shift framing away from UA being a tool for reducing impacts of urban
624 food consumption. Such research requires estimates of the current diets of city inhabitants (to
625 evaluate substitution effects) (Dorr et al., 2022a), the available space for UA (Saha and
626 Eckelman, 2017), and current city-scale flows of materials such as water and organic waste
627 (Weidner and Yang, 2020).

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

642 ***Recommendations:***

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

650 4.3 Ecosystem services and positive impacts

651 *Presentation:*

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

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

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

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

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

690 The role of biodiversity in ES is foundational, as it is defined as the source of ES (McDonald
691 et al., 2013; Millennium Ecosystem Assessment, 2005), and is often used as a proxy indicator
692 for supporting ES (Cabral et al., 2017). Improved local biodiversity is perceived as an
693 important environmental benefit of UA (Sanyé-Mengual et al., 2018b) and is frequently
694 measured in the context of ES of UA (Dennis and James, 2017; Quistberg et al., 2016). This
695 benefit is not accounted for in LCA. Biodiversity impacts in LCA have been the subject of
696 methodological development for decades, and is usually framed as the impact *on* biodiversity
697 *from* land use (or other ecological damage, although most frequently land use) (Teixeira et al.,
698 2016). LCA models the upstream and downstream impacts of materials and processes on
699 biodiversity around the world, and does not consider local biodiversity (Teixeira et al., 2016).
700 Other measures are more relevant for farm-scale biodiversity impacts like species richness,
701 habitat fragmentation, habitat vulnerability, or land use intensity indicators (Frischknecht et
702 al., 2016; Pepin, 2022).

703 ***Recommendations:***

704 For practitioners looking to operationalize ES and LCA for UA, results from each method can
705 be qualitatively assessed in parallel or quantitatively through composite indicators (De Luca
706 Peña et al., 2022). For an integrated assessment, for example comparing types of UA within
707 one study, results can be integrated in a multi-criteria decision analysis (Ledesma et al.,
708 2020).

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

712 **4.4 Social benefits and life cycle sustainability assessment**

713 ***Presentation:***

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

719 Accounting for social aspects of an activity is a main issue for LCA, and social LCA (S-LCA)
720 is a promising yet nascent strategy to overcome this (UNEP/SETAC, 2009; Zimek et al.,
721 2019). Using life-cycle thinking, S-LCA tracks the social impacts of a product's life cycle. S-
722 LCA quantifies negative impacts, and therefore may not be appropriate for evaluating the
723 social benefits of UA. S-LCA databases offer data for social impacts embedded along the
724 supply chain, but the information necessary for UA is more relevant at the farm,
725 neighborhood, or city scale (Romanovska, 2019). Plus, such databases are not as
726 generalizable as large LCA databases. A strength of S-LCA is its ability to account for the
727 perspectives of multiple stakeholders, such as workers, consumers, and the local community.
728 This is especially useful to evaluate the potential for UA to address social justice issues, by
729 highlighting not just which social benefits are brought, but who they are affecting. S-LCA
730 currently lacks agreed upon social indicators, partly because they are situational and defined

731 through stakeholder engagement, making consistent methods and comparisons between
732 studies difficult (Fauzi et al., 2019). Peri et al. (2010) outlined indicators for S-LCA of green
733 roofs, including area of green roof made accessible to the public, fair salary, working hours,
734 air pollutant levels, and outside air temperature.

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

743 Social aspects of UA may be evaluated in parallel to environmental impacts from LCA rather
744 than being fully integrated into LCA. Indeed, many researchers acknowledge that LCA cannot
745 capture everything, and it is useful to complement it with other methods (De Luca Peña et al.,
746 2022; Fauzi et al., 2019). In practice, this would be most useful to compare different types of
747 UA within a study, where the same data can be collected from a set of urban farms. UA LCA
748 practitioners should strive to measure these indicators of social benefits and present them in
749 case studies, even when they are not based on a life-cycle approach.

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

758 agriculture and are under development for UA (Clerino and Fargue-Lelièvre, 2020; Hély and
759 Antoni, 2019).

760 ***Recommendations:***

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

768 **5 Conclusion**

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

781 By applying these guidelines and strengthening UA LCAs, this research topic can better
782 support environmental sustainability of UA and cities. This research can better inform policy
783 makers about how UA implementation will affect environmental performance of cities, and

784 which types or characteristics of UA to leverage for specific goals. It can inform urban
785 farmers on how to operate or design their farms to minimize environmental impacts. They can
786 better understand which changes to implement, and which ones may not be worth the effort
787 given small environmental gains. Finally, the research community can explore methods to
788 enhance the use of LCA for multifunctional, complex activities, such as UA.

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