Circular Economy Principles in Urban Agri-food Systems: Potentials and Implications for Environmental Sustainability

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ABSTRACT

Due to urban population growth during recent decades, the food supply chain has become one of the key material flows in the metabolism of cities. Urban agriculture (UA) can be an alternative for mitigating food supply impacts. UA can provide environmental benefits, but current concepts and strategies do not reflect its full potential. The circular economy (CE) can contribute to this goal. The promotion of CE principles in UA can help mitigate the environmental impact generated by these systems and move toward circular agriculture, which extends the life of critical resources consumed in urban areas. However, it is important to identify whether the application of CE strategies in UA systems entails burden-shifting processes. The aim of this chapter is to outline and analyze the environmental implications of applying CE strategies in UA, such as the use of struvite, compost, rainwater harvesting or water and nutrient recirculation. We conclude that the application of CE strategies in UA systems should always include a parallel environmental assessment from a life cycle perspective to assess potential drawbacks and burden-shifting processes and to ensure that circular economy principles and sustainability goals are aligned.

1. GROWING IMPORTANCE OF URBAN AGRICULTURE SYSTEMS

For the first time, in 2007, the urban population surpassed the rural population (United Nations 2014). Despite this increase in urban population, cities only occupy 3% of the Earth's surface area (SEDAC 2016). It is thus expected that urban populations will consume a vast amount of the world's resources (Seitzinger et al. 2012). The food supply chain has been labeled one of the largest contributors to global environmental impacts because it is long and inefficient (Spiertz 2010). As an example, some of the most reported impacts of the food supply chain are related to water depletion (Vlachos and Aivazidou 2018) or greenhouse gas (GHG) emissions (Foley et al. 2011). The rising demand for food has also increased the pressures on natural resources and land availability (Schade and Pimentel 2010). The Intergovernmental Panel on Climate Change (IPCC) quantified that agriculture, forestry and land use contribute to 24% of the global GHG emissions (IPCC 2014). Data for the European Union (EU-28) show that agriculture contributes to nearly 10% of the overall GHG emissions (Eurostat 2011).

Considering both the current urban population and future urban population, it is unavoidable to allocate most of the impact of global agriculture on the food supply to cities. As cities rely on their hinterland to feed the urban population, understanding how to reduce the pressures arising from urban food consumption is critical (Lenzen and Peters 2010). Additionally, the COVID-19 pandemic exposed the fragility of urban food sovereignty and its dependence on external imports (Bakalis et al. 2020; Vittuari et al. 2021). In this sense, new approaches to mitigating these impacts focus on providing cities with fresh and local food (Brock 2008).

Urban agriculture (UA) systems are those located within or at the edge of a metropolitan or urban area (Smit et al. 2001) and, in a certain way, serve as an alternative to traditional methods of food production in cities (Specht et al. 2014). UA is a dynamic strategy that can be implemented in multiple forms, such as greenhouses, open-air systems, vertical farming or building-integrated agriculture (Llorach-Massana 2017). In addition to food supply or environmental enhancement (Smit et al. 2001), UA provides other benefits that can be classified according to the three dimensions of sustainability: economic, social and environmental (Specht et al. 2014; Thomaier et al. 2015).

From an economic standpoint, UA provides fresh food on demand to local markets (Despommier 2013). Moreover, UA has the power to contribute to the development of local economies (Lovell 2010; De Zeeuw 2011; Kortright and Wakefield 2011). Nonetheless, the promotion of UA, especially in rooftop areas, must compete with other novel urban uses that entail a higher economic revenue, such as photovoltaic panels (Thomaier et al. 2015).

From a social perspective, UA increases the resilience of food supply chains, contributing to food security in urban regions (Mok et al. 2014). According to Zezza and Tasciotti (2010) and Müller and Sukhdev (2019), UA may be associated with a more diverse diet, greater calorie availability and healthier food provision. Additionally, UA can increase the feeling of belonging in urban areas, create new jobs and promote social equality (Orsini et al. 2013; Ferreira et al. 2018).

The environmental perspective of UA is related to concepts such as urban resilience (Barthel and Isendahl 2013), ecosystem services (Artmann et al. 2018; Orsini et al. 2020), release of pressure to rural areas (Specht et al. 2014), reduction in transport distances (Jones 2002), sustainable cities (Taylor et al. 2012), resilience to climate change or the need to reduce water demand (Lin et al. 2015; Kalantari et al. 2018).

2. POTENTIAL OF THE CIRCULAR ECONOMY IN URBAN AGRI-FOOD SYSTEMS

Urban agricultural systems are characterized as highly resource intensive, especially in terms of land (Seitzinger et al. 2012), water (Parada et al. 2021b), nutrients (Sanjuan-Delmás et al. 2020) and substrate materials (Parada et al. 2021a). The substitution of traditional UA by innovative, circular UA systems (Figure 1) can ameliorate competition for land and resource uses in urban areas (Sanyé-Mengual et al. 2019; Rufí-Salís et al. 2021). In terms of nutrients, the linear nature of agricultural systems causes the loss of nutrients that are not assimilated by plants, either to be treated by wastewater removal technologies or to be percolated into water bodies (Garcia-Caparros et al. 2017; Rufí-Salís et al. 2020b).

Apart from the potential impacts related to eutrophication caused by nitrogen (N) and phosphorus (P) emissions, one of the main concerns of intensive agriculture is the inefficiency of P resources (Cordell et al. 2009; Rufí-Salís 2020). Currently, P is primarily obtained from phosphate rocks, with 80% of the available stock employed in the production of fertilizers (Shu et al. 2006). Considering that P deposits are renewed on a scale of geologic time (Childers et al. 2011) and that unlike N, P does not have a mechanism to move from marine stocks to terrestrial stocks (Elser and Bennett 2011), mineral P resources are likely to be depleted in the 21st century (Steen 1998). Thus, given the scientific consensus on the depleting nature of P (Rittmann et al. 2011) and the risks that this depletion poses to food systems, sustaining P reserves has already reached the political agenda. For instance, in the EU, phosphate rock is considered a critical raw material (European Commission 2020a). In the period 2012-2016, the EU imported approximately 85% of phosphate rock (in P_2O_5 content) mainly from Morocco (28%), the Russian Federation (23%) and Algeria (13%) (European Commission 2020b). The supply dependence in the EU on producing food for cities could be alleviated by the use of secondary sources of P, mainly from animal manure, food waste and wastewater (van Dijk et al. 2016).

In this sense, the application of CE principles can be a path to increase the efficiency of critical resources and to decrease the depletion of critical resources that are utilized in UA systems and the systems that can generate symbiosis at an urban or regional level (Figure 1). As stated by Ferreira et al. (2018), “*agriculture is central to any territorial-based circular economy strategy*”. Along general lines and considering the EU context, Figure 1 shows how the application of CE principles in UA systems is also a good strategy to reduce the import dependency of materials, such as P resources, from developing economies and to improve the internal EU market by moving away from linear agricultural production that consumes primary inputs and generates waste. This finding translates to a reduction in the dependency on external resources, especially regarding water as the main input of UA systems (Rufí-Salís et al. 2021). As exemplified in Figure 1, moving from circular agriculture without a defined scale to circular agriculture that accounts for urban metabolism principles can ameliorate resource flows, such as virtual water, and the externalization of impacts at the expense of increasing locally reused water. However, we need

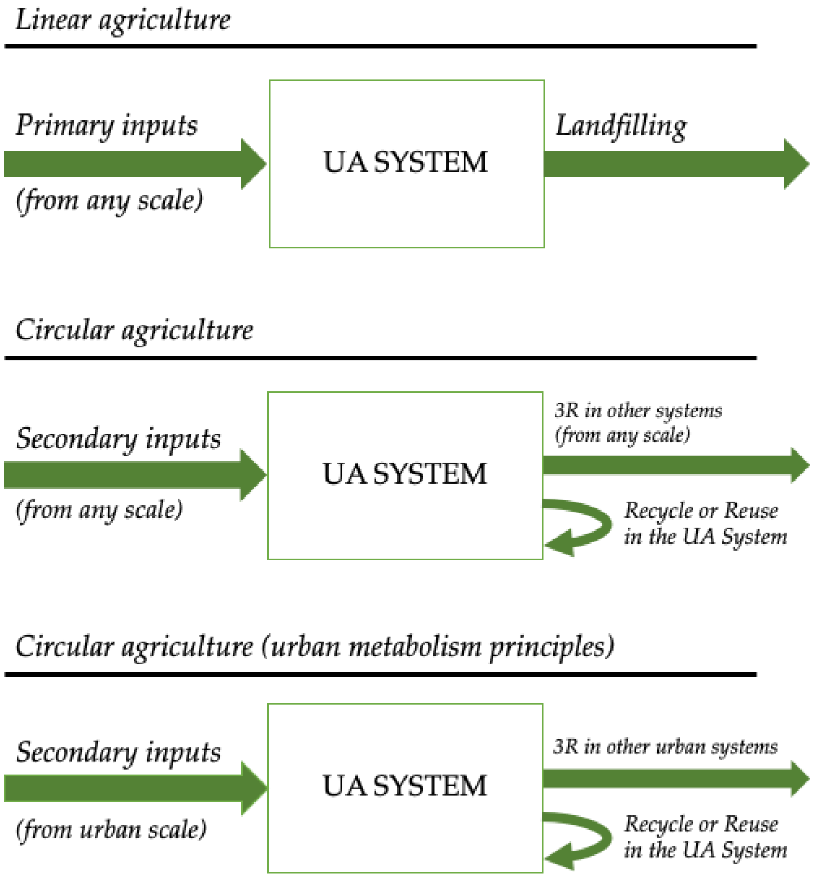

Figure 1. Differences between linear and circular agriculture (with or without urban metabolism principles). Primary inputs refer to the use of virgin and raw materials, while secondary inputs refer to the use of recovered resources to which a second life is being given. 3R: recycle, reuse, revalorize; UA: Urban Agriculture.

specific strategies that focus on the enhancement of the UA system at different scales (Figure 2).

3. CIRCULAR ECONOMY AS A MEAN – A LIFE CYCLE PERSPECTIVE

Although the application of CE principles to UA systems may have apparent benefits, such as increasing the efficiency of water and nutrient flows and reducing direct emissions and wastes, to what extent these benefits compensate for potential trade-offs should be quantified. This aim requires a broad environmental analysis of not only all the elements of the UA system under assessment but also all the life cycle stages associated with each of the UA elements involved: extraction of raw materials, processing, or end-of-life, among others. The use of life cycle assessment (LCA) can help to quantify to what extent the CE is a means to improve environmental sustainability (ISO 2006; Peña et al. 2020), since previous studies have indicated that closing loops is not always a means to improve environmental performance (e.g. Laner and Rechberger, 2007; Humbert et al., 2009; Geyer et al., 2016; Rufí-Salís et al., 2020e). The application of LCA principles is vital to detecting weaknesses and environmental hotspots of CE strategies not only at the end of life through a redefinition of the waste hierarchy but also among different stages of the life cycle of a system. In this sense, a life cycle perspective is mentioned in the recent CE

Action Plan from the European Union: "*this legislative initiative […] will be developed in a way to improve the coherence with existing instruments regulating products along various phases of their life cycle*" (European Commission 2020c). As expressed in the last Life Cycle Initiative's position paper titled "*Using Life Cycle Assessment to achieve a circular economy*", LCA should be applied as a methodology to promote more robust circular strategies that include all relevant resources and indicators, leading to better decisions for sustainability (Peña et al., 2020). Additionally, Zeller et al. (2019) states that "*whether the closing of material cycles at city level has environmental benefits compared to the national or global level, it needs to be further assessed based on comparative LCA studies.*"

4. BENEFITS AND TRADE-OFFS: A SERIES OF CASES

Given the potential of UA as an opportunity to utilize wastes as resources within city limits (Smit et al. 2001; Ferreira et al. 2018), the application of CE strategies in UA systems is a promising path toward not only circular UA systems but also a more circular urban metabolism (Figure 1). In this section, we describe some of the most common CE strategies mentioned in the literature

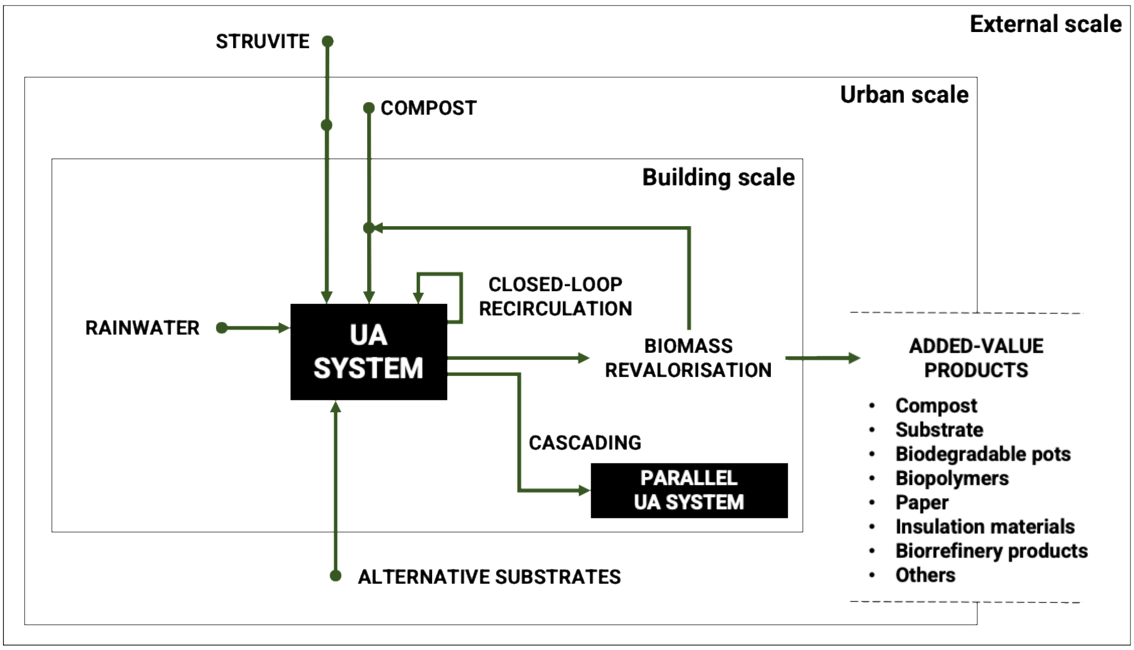

Figure 2. Circular Economy strategies discussed in the present chapter. Adapted from Rufí-Salís et al. (2021).

and discuss their potential benefits and trade-offs in terms of environmental sustainability. Figure 2 shows the strategies that will be described in the following subsections and classifies them into the scales on which they would be feasible. If a strategy is classified under the "building scale", it could be operational by using secondary inputs in the same building, including both the UA system itself and other waste-to-resource processes, such as organic waste generation. If a strategy is classified under the “urban scale”, it could be operational using secondary inputs generated in systems that are often identified at a generic urban scale (including both city limits and urban/metropolitan region limits), such as municipal solid waste collectors and processors or wastewater treatment plants. Any strategy included in the “external scale” may not be feasible at the building and urban scales for a variety of factors, such as the nonexistence of recovery technologies in that specific city or the scarcity of secondary inputs.

4.1 Rainwater harvesting

The use of harvested rainwater in the urban context can alleviate the limited water availability in these areas (Toboso-Chavero et al. 2018), especially for semiarid urban territories, where it can cause a significant reduction in water consumption, marine eutrophication and freshwater eutrophication impacts (Bonilla-Gámez et al. 2021). The use of rainwater at the building scale is a strategy that has been extensively investigated and is able to provide nearly 100% of the water for hydroponic cultivation (Sanjuan-Delmás et al. 2018; Parada et al. 2021b). However, concerns have been raised in the literature regarding the potential life cycle impacts across multiple impact categories (e.g., global warming or resource use) exerted by storage tanks in their extraction and manufacturing phases (Sanjuan-Delmás et al. 2018). In this sense, the parameter that can be modified is the size of the tank, accounting for local pluviometry conditions and water requirements of the urban agricultural system (Mun and Han 2012; Angrill et al. 2012). Software programs such as Plugrisost© are already available to model the decreased impact of rainwater harvesting systems while maintaining the benefit of reusing water (Gabarrell et al. 2014).

4.2 Closed-loop Hydroponic Cultivation

The use of hydroponic cultivation in rooftop farming intrinsically improves the water and nutrient supply efficiency by allowing for better control of plant nutrition (Christie 2014). Moreover, hydroponic cultivation also allows precise monitoring of the leachates and increases the flexibility of the UA system in managing the residual water and nutrient flows (Sayara et al. 2016). In this sense, different strategies are discussed in the literature to take advantage of these residual flows generated at a building scale. The recovery of valuable nutrients, such as P compounds, is being investigated and improved through different methods, such as chemical precipitation or membrane filtration (Rufí-Salís et al. 2020c). However, these technologies may require high investment costs, so they are only feasible given a considerable high residual flow (only reachable in large facilities) or by sending the residual flows to urban wastewater treatment plants. Accordingly, these technologies are usually related only to the removal of nutrients, with the principal goals of cleaning the water before losing it to water bodies and of complying with legal standards. Additionally, removal efficiency is still highly dependent on the degree of technological innovation (e.g. Piekema and Giesen, 2001; Tchobanoglous et al., 2003; Royal Haskoning DHV, 2019).

To ensure that all water and nutrients are recovered, the use of closed-loop hydroponic cultivation is reported in the literature (Agung Putra and Yuliando 2015; Bouchaaba et al. 2015), although limitations and potential trade-offs are also mentioned from studies that quantify environmental impacts. The recirculation of the drainage effluent in the same crop (thus classifying this strategy on a building scale) not only mitigates the eutrophication impacts from direct emissions but also directly diminishes the required inputs to agri-food systems (Parada et al. 2021b). However, potential limitations may arise if nutrient metabolism is not included in the analysis (Rufí-Salís et al. 2020e), since imbalances can negatively affect plant development and increase the environmental impacts due to a decrease in the quantity of the functional unit. Therefore, it is highly relevant to measure all nutrient flows within the system to maximize its efficiency. The use of drainage effluent in parallel crops in what are referred to as cascade systems is a potential way to overcome this limitation (Incrocci et al. 2003; García-Caparrós et al. 2016; Rufí-Salís et al. 2020d). Nonetheless, the largest trade-off detected is related to a potential burden-shifting in terms of infrastructure if the materials needed to implement closed-loop systems have a high environmental contribution (Rufí-Salís et al. 2020e; Parada et al. 2021b). This finding highlights the need for an extension of the assessment boundaries of CE strategies, sometimes narrowed at the end of life, to production and manufacturing considering eco-design principles, low-impact alternatives, and potential burden-shifting processes across life cycle stages.

4.3 Use of recovered resources: struvite as a secondary fertilizer

As mentioned earlier in this chapter, phosphate rock, one of the main mineral sources of P, is considered a critical raw material in the EU. Identifying secondary sources of P, such as struvite, is a priority to make agriculture as self-reliant as possible. Additionally, the use of secondary fertilizers directly contributes to circular urban metabolism since it links two potentially relevant sources of impact in cities: wastewater treatment and agricultural production. Returning to the P case, a great example of industrial symbiosis in urban areas is struvite (Talboys et al. 2016). Struvite is a mineral composed of magnesium, phosphate, and ammonium in a 1:1:1 stoichiometric relationship that is accidentally precipitated in the ducts of wastewater treatment plants. Technologies aimed at removing struvite through induced precipitation to diminish maintenance and repair costs in wastewater treatment plants offer an unexpected opportunity. Struvite has been extensively tested in the literature, with results suggesting that higher production can be reached at a lower environmental cost (Linderholm et al. 2012; Amann et al. 2018).

The recovery of struvite in urban wastewater treatment plants and its reuse in urban and periurban agricultural production has already been quantified to be feasible at a regional scale (Rufí-Salís et al. 2020a), although P-recovery technologies should be implemented to avoid relying on struvite imports from other regions (External Scale – Figure 2). Thus, the next challenge is to quantify to what extent a regional CE plan for P is feasible in other key urban areas and to explore to which other linear flows the successful case of P can be extrapolated.

4.4 Use of recovered resources: alternative substrates

The use of stabilized compost from municipal solid waste, which comprises a tremendous flow at the building and urban scales, can enhance the resilience of nutrient supply chains and contribute to prioritizing composting processes over alternatives with less added value, such as incineration or cogeneration. The use of compost as a fertilizer can enhance production and have lower environmental impacts than commercial alternatives (Martínez-Blanco et al. 2009, 2011).

Apart from the use of compost as a nutrient supplier, recent studies have explored the use of this material as a potential substrate to displace primary materials such as perlite or rockwool, which are traditionally employed in hydroponic agriculture (Olle et al. 2012). Apart from being available from local markets, compost contributes to closing loops at a smaller scale, while the production of commercial alternatives is usually concentrated in specific countries, such as perlite in Turkey or Greece (USGS 2013). Nonetheless, CE strategies that are aimed at replacing conventional materials must always consider local characteristics. For example, hydroponic settings in rural areas may not have a large market for municipal solid waste as urban areas. Therefore, local alternatives such as sheep wool, dried moss, pine bark or wood fiber can constitute a more resilient and reliable market for substrate provision (Barrett et al. 2016). In this sense, previous literature has demonstrated that dried moss might be more agronomically efficient than sheep wool (Dannehl et al. 2015). Therefore, it is also important to account for previous experimental tests and findings in the academic literature to plan the implementation of CE strategies in a specific region or agricultural system. As an example, Parada et al. (2021) discovered that compost was a viable alternative as a hydroponic substrate, but its mixture with perlite provided more tolerance to temporary water restriction than perlite as a standalone substrate. This finding may be important in regions where compost and water are limited. Moreover, safety in use, for example, in terms of heavy metal content (Ercilla-Montserrat et al. 2018), is crucial to increase the acceptability of alternatives both to farmers and consumers.

4.5 Added-value secondary products from urban agriculture

Considering the increasing spread of UA systems, insights into the main waste flows exiting these systems are critical to ensure that they contribute to a more circular urban metabolism. However, information about the quantification of waste flows from UA systems is lacking. A characterization of UA solid waste in the framework of the CE was performed by Manríquez-Altamirano et al. (2020). This study, which is based on a hydroponic system in a rooftop greenhouse, determined that each m^2 of crop annually generates 2.4 kg of inorganic waste and 6.6 kg of organic waste. Within this classification, inorganic outputs were mainly composed of substrate waste, while organic outputs were composed of branches and leaves (77%) and stems (23%).

Although inorganic waste represents 3.5 times the amount of organic waste, previous research that analyzes potential uses from the waste generated in UA has focused on organic outputs, more specifically on biomass reuse and recycling. The academic literature highlights the use of biomass waste as a substrate (Grard et al. 2018) and compost (Goldstein et al. 2016). However, recent studies have explored the use of biomass to create biobased, added-value products (refer to Figure 2). For example, Manríquez-Altamirano et al. (2021) discussed with a panel of experts three potential groups of biobased applications from tomato stems: fences, packaging and boards, panels and blocks. Parallel research on recycling other parts of tomato plants has demonstrated the potential to create other circular bioeconomic products, such as biodegradable pots (Schettini et al. 2013) or biopolymers (Franzoso et al. 2015). By mixing tomato stems with other biobased materials, previous literature evaluated the production of paper (Üner et al. 2016) or insulation materials (Llorach-Massana 2017).

5. CIRCULARITY ASSESSMENT OF URBAN AGRI-FOOD SYSTEMS: HOW TO LINK IT WITH ENVIRONMENTAL PERFORMANCE

Unlike LCA, which is standardized through an ISO document (ISO 2006), the methodology for determining to what extent a system is indeed circular is still under discussion. Several indicators have already been reported in previous research, each of them with their own potentials and limitations. To ease the selection, reviews of indicators are provided in the academic literature (e.g. Haupt et al., 2017; Helander et al., 2019; Saidani et al., 2019).

Not all indicators can be applied in all situations. Since a systems perspective is intrinsically associated with UA systems, scaling up indicators that are focused on products may entail a certain degree of uncertainty. Considering this point, reliable indicators for assessing the circularity of the agri-food sector are still lacking (Kristensen and Mosgaard 2020). In UA, where most of the foreground system is based on biological materials, an analysis of circularity may focus only on closing loops at the end-of-life stage via biological processes (i.e., composting and biodegradation). Other strategies, such as lifetime extension, repair, and remanufacture, may not be applicable to the foreground system of UA, especially if it is narrowed from an operational approach. However, if we want to link the circularity assessment with an analysis of environmental performance, the system boundaries must be comparable. As highlighted by Helander et al. (2019), the Material Circularity Indicator (MCI), which was developed by the Ellen MacArthur Foundation (EMF 2015), is applied across life cycle phases, rendering it complementary to LCA. In this sense, a dual assessment that combines LCA and MCI results has already been performed for UA by Rufí-Salís et al. (2021) but also for packaging by Niero and Kalbar (2019). These papers present different ways of coupling the MCI and LCA, either by multicriteria decision analysis to obtain a single factor or by developing a new set of indicators through environmental relative contributions. Apart from the potential discussion on the best way to couple circularity and LCA indicators, several limitations were reported.

Carbon fixed by short-cycle crops (labeled biogenic) or nutrient emissions to water without eutrophication potential are part of the flows that are not usually collected in LCA studies. However, these flows must be quantified in circularity assessments since they encompass mass conservation of all flows. Additionally, challenges may arise in assigning the terminology required for the MCI, such as virgin feedstock or unrecoverable waste, to specific flows (e.g., drained water or solid waste for which the next use is unknown).

Another potential limitation is related to the water flow. Water may hinder the results by representing most of the MCI total value, which is also reported by economy-wide material flows analysis (Eurostat 2001) and thus limits the assessment of additional flows with less mass. Other limitations highlighted include the omission of transport or energy processes (Rufí-Salís et al., 2021; Saidani et al., 2019a). Based on the complexity of integrating LCA and circularity indicators highlighted by previous attempts, maintaining the raw values obtained either through LCA or circularity assessment seems to be the safest and least uncertain way to proceed while standardization procedures are under construction.

6. IMPORTANCE OF GEOGRAPHICAL SCALES

The application of CE principles is not a standalone objective but a potential means to increase a system's overall sustainability. Considering the three sustainability dimensions, locating circular strategies within a framework of different geographical levels can help contextualize broader implications than those captured by circularity indicators. As an example, Rufí-Salís et al. (2021) quantified 13 different circular strategies that could be applied to a hydroponic rooftop greenhouse, classified within building, urban and national scales. Within the circular strategies analyzed, the use of compost as a potential substrate showcased how a geographical perspective may be important in quantifying the environmental sustainability of circular strategies. The use of compost presented limitations at the building scale due to the amount of biowaste produced but was feasible at the urban scale because of the amount of compost produced from municipal solid waste. Another potential example of the importance of a scale framework from the same study is the case of struvite. Although the potential struvite obtained from urban wastewater treatment plants has been demonstrated to be feasible at a regional scale for the Barcelona metropolitan area (Rufí-Salís et al. 2020a), the amount of struvite that can be obtained at the crop level renders it an inefficient method compared with other CE alternatives, such as direct recirculation (Rufí-Salís et al. 2020b). Thus, the conclusion is that the geographical scale needs to be considered to assess the feasibility of the development and implementation of circular economy practices.

7. IDENTIFYING AND ADDRESSING ENVIRONMENTAL BURDEN-SHIFTING PROCESSES

From a methodological standpoint, burden shifting represents the major challenge in the application of CE strategies in UA systems from an environmental perspective. In this sense, LCA must be performed in a way that avoids solving one problem at the expense of another problem (Brandao et al. 2017). Three potential burden-shifting processes were identified in this chapter: between impact categories, between life cycle stages, and between systems, markets or generations.

Burden shifting between two impact categories was demonstrated to be quantified through LCA (European Commission 2013). For example, recirculation systems, one the most common CE strategies in the literature, can potentially reduce the eutrophication potential of the system by avoiding nutrient discharge but may contribute to increasing the global warming potential due to additional materials required to adapt the system setup (Rufí-Salís et al. 2020e). The use of LCA is useful for detecting these weaknesses (Peña et al. 2020). However, LCA must be performed not only to quantify the environmental hotspots of the system in specific impact categories but also to design mitigation strategies (ISO 2006).

Burden-shifting between two life cycle stages is also quantified through LCA (Hauschild et al. 2017). As stated by ISO (2006), "*through such a systematic overview and perspective, the shifting of a potential environmental burden between life cycle stages or individual processes can be identified and possibly avoided*". However, the omission of specific life cycle stages in the assessment either due to a lack of data or low impact in the baseline scenario can decrease the efficiency of LCA in detecting these kinds of problems. To avoid this situation, it is critical that LCA accounts for all the impacts occurring throughout the entire value chain (Hellweg and Canals 2014). Using the previous example, if the study from Rufí-Salís et al. (2020e) had only accounted for the operational impacts of applying a closed-loop strategy (i.e., use and maintenance of the system), the potential environmental impacts from the manufacturing of the additional infrastructure required for the recirculation system would have been hidden in a blind spot. Therefore, the environmental assessment of the application of CE strategies should encompass all the life cycle stages and should not be limited to only the relevant life cycle stages for the baseline scenario, since unexpected hotspots may appear along the supply chain of the different inputs and outputs linked with UA systems.

Burden-shifting can also occur in terms of spatial and temporal resolution (European Commission 2013). However, burden-shifting processes among systems, markets or generations seem unlikely to be detected from an attributional approach that establishes the system boundaries at the system level. The use of LCA from an attributional approach narrows the direct effects of a CE strategy to the system to which it is applied and the flows into and out of the system. In this sense, Schmidt (2008) states that modeling agricultural LCA from an attributional approach produces some blind spots. As highlighted by previous literature, avoiding the consequences from a market perspective, such as price fluctuations or rebound effects, can cause burden shifting (Zink and Geyer 2017). In this sense, the use of consequential LCA (see Weidema (1993) and Zamagni et al. (2012)), which focuses on the consequences generated by a change in demand of the functional unit) has been labeled a powerful tool to avoid burden shifting along the different supply chains linked to agricultural LCAs.

In a research paper on the mitigation potential of CE principles, Cantzler et al. (2020) highlights that a consequential approach is preferred over an attributional approach when communicating with policy makers. Taking into account that the CE has now reached the political agenda at different levels (e.g. People's Republic of China, 2008; The White House, 2012; European Commission, 2020), identifying the most accurate metrics to transition from linear behavior to CE strategies will continue to be a hot topic for a broad variety of stakeholders.

8. FINAL REMARKS AND UPCOMING CHALLENGES

This chapter has presented a summary of the current CE strategies applied to UA systems discussed in the literature and their contribution to environmental sustainability. From a system's perspective, three main challenges can be drawn from the analysis.

Standardization of CE definitions and metrics: not only the variability of CE definitions available in the literature (Kirchherr et al. 2017) but also the number of methodologies to measure circularity (e.g. Haupt et al., 2017; Helander et al., 2019; Moraga et al., 2019) makes it extremely difficult to compare the current works on these novel topics. Therefore, it is urgent to conceptualize a CE definition and the ways to measure progress toward it in every specific system.

With a clear definition, the path to determine to what extent the application of CE principles in systems such as UA can contribute to environmental sustainability will be clearly defined for practitioners, decision-makers and all relevant stakeholders. The development of the currently on-going ISO standards on the circular economy by the ISO/TC323 would contribute to standardizing definitions and methods and therefore advance the assessment of current and novel strategies within systems (ISO 2018).

Accounting for all parameters that have a role: the environmental assessment of CE strategies in UA systems should account for not only all the relevant impacting items but also those parameters that affect the performance of the system, such as the nutrient metabolism or climatic conditions, to detect potential nutritional deficiencies or excessive evapotranspiration. When considering many parameters, the results might not be conclusive, but having more detailed information about the effects of changes in certain parameters would yield better decisions.

A system's perspective on the implementation of CE strategies and its assessment: applying a system's perspective to the implementation of CE strategies encompasses good knowledge about the system from the actor evaluating it: relevant flows, factors affecting its behavior, level of resilience, etc. A CE strategy that may improve the environmental performance in hydroponic systems might not be that efficient for soil-based setups. A similar situation can happen between open production systems and greenhouse production systems and with a great variety of different factors. Therefore, an in-depth analysis of how a system is designed and works on a daily basis is critical to planning the most suitable CE strategies not only for UA but also for all other assessable systems.

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