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Environmental impacts and resource use of urban agriculture: a systematic review and meta-analysis

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ENVIRONMENTAL RESEARCH
LETTERS

Environmental impacts and resource use of urban agriculture: a systematic review and meta-analysis

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Abstract

Environmental merits are a common motivation for many urban agriculture (UA) projects. One powerful way of quantifying environmental impacts is with life cycle assessment (LCA): a method that estimates the environmental impacts of producing, using, and disposing of a good. LCAs of UA have proliferated in recent years, evaluating a diverse range of UA systems and generating mixed conclusions about their environmental performance. To clarify the varied literature, we performed a systematic review of LCAs of UA to answer the following questions: What is the scope of available LCAs of UA (geographic, crop choice, system type)? What is the environmental performance and resource intensity of diverse forms of UA? How have these LCAs been done, and does the quality and consistency allow the evidence to support decision making? We searched for original, peer-reviewed LCAs of agricultural production at UA systems, and selected and evaluated 47 papers fitting our analysis criteria, covering 88 different farms and 259 production systems. Focusing on yield, water consumption, greenhouse gas emissions, and cumulative energy demand, using functional units based on mass of crops grown and land occupied, we found a wide range of results. We summarized baseline ranges, identified trends across UA profiles, and highlighted the most impactful parts of different systems. There were examples of all types of systems—across physical set up, crop type, and socio-economic orientation—achieving low and high impacts and yields, and performing better or worse than conventional agriculture. However, issues with the quality and consistency of the LCAs, the use of conventional agriculture data in UA settings, and the high variability in their results prevented us from drawing definitive conclusions about the environmental impacts and resource use of UA. We provided guidelines for improving LCAs of UA, and make a strong case that more research on this topic is necessary to improve our understanding of the environmental impacts and benefits of UA.

1. Introduction

Urban agriculture (UA) is generally considered to be an environmentally sustainable activity, with low impacts and numerous benefits. It is often defined by its location in or around an urban area, and by its material and human links with the city, although specific characteristics of cities can mean that the application of this definition varies (Mougeot 2000).

It is broadly asserted that UA is a resource-efficient form of sustainable agriculture that can contribute to climate change mitigation (McEldowney 2017, Artmann and Sartison 2018, Feola *et al* 2020). Specific mechanisms for achieving this have also been proposed, either theoretically or demonstrated with disparate case studies, and include reduced transport for distribution, reduced food waste along the distribution chain, reusing urban waste as an input,

integration with buildings to reuse waste heat and rainwater runoff, employing agroecological practices, shifts towards more environmentally conscious habits by participants, among others (Specht *et al* 2014, Goldstein *et al* 2016a, Grard *et al* 2018, Alemu and Grebitus 2020, Orsini 2020, Dobson *et al* 2021). At the same time, opposing results have been found in specific case studies, suggesting that these benefits can be overstated. Indeed, studies have shown that UA can have larger climate change impacts than its conventional counterpart (i.e. rural agriculture), and that it can use resources inefficiently (Goldstein *et al* 2016b, McDougall *et al* 2019, Tharrey *et al* 2020). The so-far mixed evidence has come from individual case studies which used multiple methods and have not been summarized, preventing us from drawing trends from this research.

Clarifying our understanding to draw such trends, plus evaluating the quality of this evidence, is timely and valuable. Such work can support policy makers, and urban farmers and gardeners themselves, in making decisions based on the actual performance of UA, emphasizing or demoting some environmental justifications for UA, optimizing these systems, and envisioning the consequences of scaling up UA in cities. Indeed, UA is directly promoted worldwide, evidenced by its inclusion in Milan Urban Food Policy Pact, which over 200 cities have signed (Milan Urban Food Policy Pact, 2015), and the European Union Biodiversity Strategy for 2030 (European Commission 2020). Plus, it is indirectly supported by many programs that promote sustainable agriculture and cities, such as the United Nation's Sustainable Development Goals (United Nations 2015). This work is also relevant for farmers and gardeners, for whom environmental concerns can be a top motivator (Guitart *et al* 2015, McDougall *et al* 2019, Siegner *et al* 2020).
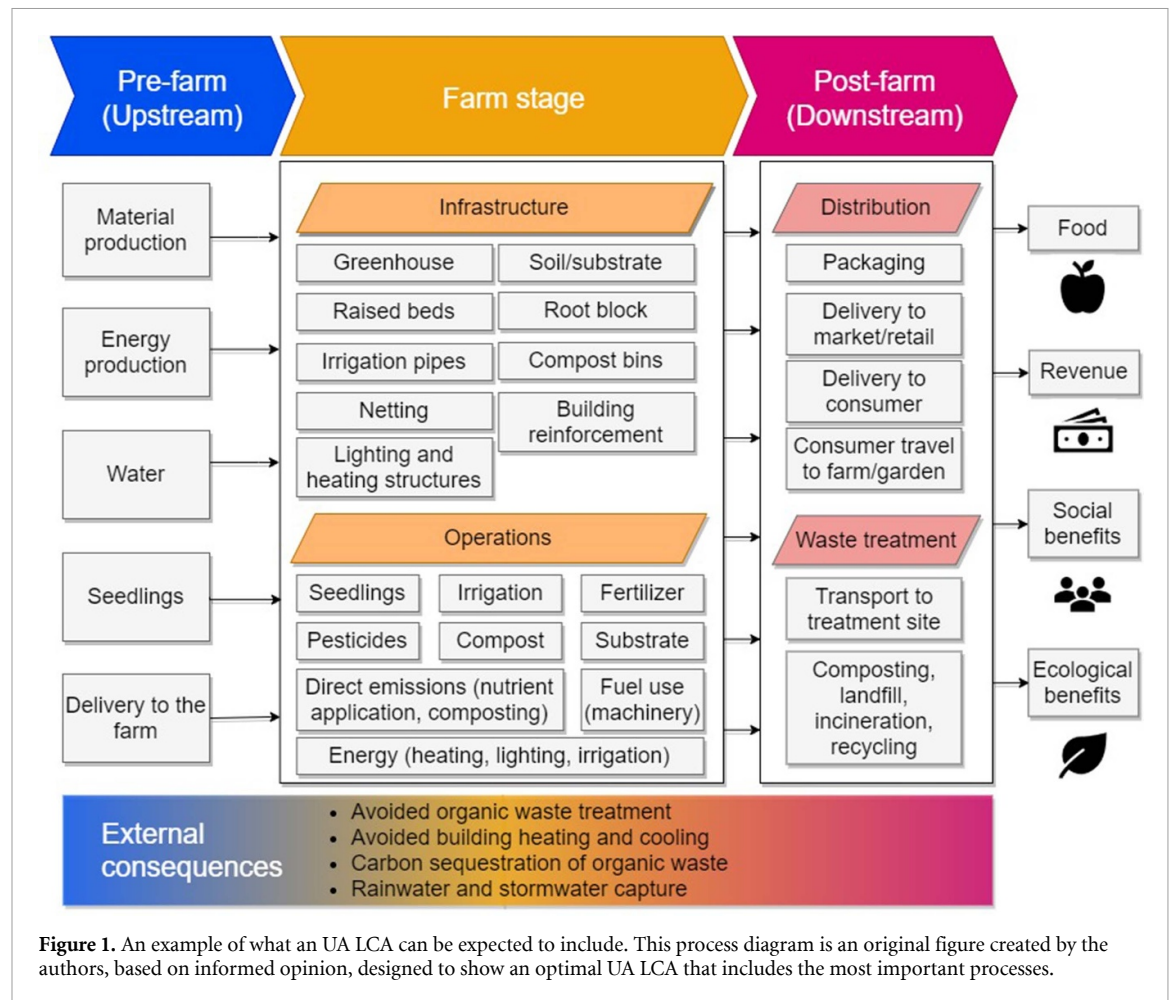
A powerful method for measuring environmental impacts of such production systems is life cycle assessment (LCA). The life-cycle perspective takes into account not only activities at the farm or garden, but also 'upstream' (pre-farm) and 'downstream' (post-farm) activities off the farm, and embodied impacts in materials used at the farm (ISO 14040 2006). This is often called the 'cradle to grave' perspective, because everything is included from the extraction of raw materials to the final waste treatment of the product. Environmental impacts of these activities are modeled and summed across the life-cycle of a product, and normalized to an output of the system: either a product or a service. Therefore, LCAs allow for comparing impacts of complex systems with the same functions and outputs. Another valuable use of LCA is identifying 'hot spots' in a system, meaning the parts of the life-cycle that have the largest environmental impacts. LCA has been historically and predominantly used for technical manufacturing systems (e.g. concrete and cars), and was only

applied to food and agriculture production decades later (Audsley *et al* 1997, Haas *et al* 2000). The method is less developed for use in agriculture systems, but there are still extensive reviews of LCAs of conventional agriculture products, food types, and production methods (Poore and Nemecek 2018, Sala *et al* 2021). The UA community lacks such reviews clarifying the methods and results of LCA of UA.

1.1. The ideal urban agriculture life-cycle assessment

LCA can be used to evaluate UA at multiple scales (farm, consumer, urban food system…), but the foundation for any of these is an appropriate evaluation at the farm. This is the scale that is most often studied currently (Fisher 2014, Martin and Molin 2019), and once it is better understood, results can reliably feed models for scaled-up UA impacts, as has been done for conventional agriculture. As with all LCAs, data should be used that accurately reflect the system being studied. Therefore, for an ideal UA LCA, data should be measured (or less ideally, estimated) from urban farms or gardens, or otherwise taken from similar UA systems (although that has frequently not been done for UA LCAs). For generalizable results, systems should be chosen and studied that are representative of UA in a city or area. Alternatively, unique and innovative systems are useful to study, given that their novelty status is made clear.

Using a life-cycle perspective to evaluate an urban farm or garden means that pre-farm and post-farm systems should be included when calculating environmental impacts. Figure 1 was created by the authors, based on informed opinion, to show key upstream, on-farm, and downstream elements of an urban farm or garden that are expected to be included in an optimal UA LCA. Upstream processes include production of inputs to the farm, including materials (e.g. fertilizer), water, and energy, plus the transport of materials to the farm. This is especially important for long-distance transport, and high-frequency deliveries across short distances. The farm-stage includes the use of inputs, and is mostly composed of embodied impacts, although some direct impacts here include nitrogen emissions from nutrient application. Downstream processes include two major categories: waste treatment and, where relevant, distribution of the product. Waste treatment should cover inputs such as infrastructure waste at the end of its life, waste of consumed inputs with shorter lifespans, such as pots, edible food waste, and residual plant biomass waste (considering that for example for every kilogram tomato produced, 0.31, 0.44, or 0.94 kg of non-edible biomass are produced as a by— or waste-product (Sanjuan-Delmás *et al* 2018, Boneta *et al* 2019, Manríquez-Altamirano *et al* 2020)). For each waste material, we can consider the steps of collection and transport to the waste treatment site (or not, if plant biomass is composted on-site), and then

Figure 1. An example of what an UA LCA can be expected to include. This process diagram is an original figure created by the authors, based on informed opinion, designed to show an optimal UA LCA that includes the most important processes.

the actual treatment of the waste, through landfilling, incineration, recycling, composting, or other techniques. Distribution includes packaging and transport to the consumer or retail market. It is useful to include this in a UA LCA even if no packaging or transport is necessary—for example, when volunteer gardeners take produce home or it is delivered on foot. Then at least the low (zero) impact of distribution can be accounted for.

We identified a separate category of processes called 'external consequences', which can occur anywhere through the life-cycle. This includes various processes or avoided processes that are often justified as direct consequences of a UA system, and are credited within the system boundary of a farm or garden. This can include avoided municipal organic waste treatment if organic waste is diverted to the farm or garden to make compost, or avoided heating of a building where a rooftop greenhouse (RTG) provides insulation. These processes can have enormous effects on the final LCA results, or be relatively minor (Goldstein *et al* 2016b, Dorr *et al* 2017). They are inconsistently applied in UA LCAs because their relevance for actual systems can be varied, and researchers' decisions to credit or burden UA systems vary depending on the research goals and the context.

1.2. The reality of life-cycle assessment of urban agriculture

The current body of literature on UA LCAs uses various methods with various results, and does not consistently adhere to the ideal UA LCA framework proposed above to create a reliable foundation of evidence on UA. Many UA LCAs focus on food production at the farm level (Sanyé-Mengual *et al* 2015b), but some use the city scale (Benis and Ferrão 2017) and land-use function (Corcelli *et al* 2019). These assessments often presume that UA has reduced environmental impacts from the conventional food systems, and research is framed as quantifying these benefits and reduced impacts (Kulak *et al* 2013, Cleveland *et al* 2017). Sometimes the conclusion is that UA greatly reduces climate change impacts from conventional agriculture, when only the reductions were modeled, with no impacts from UA itself (Cleveland *et al* 2017, Vávra *et al* 2018). Sometimes data come from specific, functioning urban farms and gardens (Fisher 2014), and sometimes data come from the scientific literature or models (Weidner and Yang 2020). Diverse forms of UA have been studied, including community gardens (Emery and Brown 2016), research farms (Dorr *et al* 2017), low-input school gardens (Ledesma *et al* 2020) and high-tech

indoor vertical farms (Martin and Molin 2019), all with varying relevant processes and external consequences. A critical review of this literature is necessary to assess the consistency and quality of these LCAs in order to evaluate the strength of the literature towards supporting conclusions and decision making about UA. In a second step, the outcomes of these disparate LCAs need to be summarized, to show what the available evidence says about environmental impacts of UA.

1.3. Study aims
Given the recent emergence and accumulation of UA LCAs, and the current relevance of UA in many cities, a first systematic review and meta-analysis of the topic is necessary. The principal goal of this review is to summarize how LCA has been applied to UA in the academic literature thus far and the outcomes of these studies, and to evaluate their quality in terms of methodology and data. Towards that end, we ask the following three questions:

(a) What is the scope of UA LCAs in terms of types of systems assessed, crops and geographical areas?
(b) What is the environmental performance and resource intensity of diverse forms of UA?
(c) How have researchers been performing LCAs of UA, and, based on the quality and consistency of the UA LCAs available, to what extent can the literature support conclusions about UA?

Through this review, we hope to summarize the findings and relevance of the available literature, and provide a roadmap for how to better perform LCAs of UA to advance our understanding of the environmental performance of UA.

2. Methods

This review followed the standardized technique for assessing and reporting reviews of LCA (STARR–LCA, largely based on the PRISMA format), including the description of review protocol below (Zumsteg *et al* 2012).

2.1. Search and selection criteria
We searched online databases Web of Science and Scopus for peer-reviewed articles, conference papers, and books using simultaneously the keywords 'urban agriculture' and 'life cycle assessment'. We also included variants of these terms, such as urban garden, farm, greenhouse, hydroponics, rooftop farm, community garden, building integrated agriculture, vertical farm, and plant factory; and life cycle analysis, carbon accounting, and carbon footprint. The specific search queries and their results are provided in the supplementary material (available online at stacks.iop.org/ERL/16/093002/mmedia). The last literature search was performed in April 2021. To be included in the review, a study had to:

(a) present an original LCA;
(b) evaluate operations and agricultural production at an UA system or systems;
(c) be a peer-reviewed journal article, scholarly book chapter, or peer-reviewed conference paper;
(d) and, for inclusion in the meta-analysis, present harmonizable LCA results at the farm level, based on food output, land occupation, or total farm operation. Examples of what was excluded are described below.

Our initial search yielded 352 resources, which was reduced to 308 after removing review papers (as they are not original LCAs, plus none focused on UA and LCA) and editorials, and selecting only articles, conference papers, and book chapters. After reading the abstracts, we eliminated sources that were clearly not in urban settings or not focused on agricultural production, narrowing the literature to 132 sources. We assumed that if the title and abstract did not include the words 'urban' or 'city', or mention the name of a specific city, it was about conventional agriculture rather than UA. Finally, close reading of the full papers led us to eliminate 85 papers because they did not meet the first three criteria above, resulting in 47 papers considered for the systematic review. Nine ended up not included in the meta-analysis due to methodological differences that could not be harmonized with the other papers according to above criteria, and are described in greater detail below.

Regarding our original LCA criteria, we excluded studies that incorporated an LCA already in the published literature and applied other non-LCA methods to the analysis. For example, we did not include studies that presented a previously published LCA of UA and complemented it with socio—economic assessments or ecological network analysis. In such a case, we only included the previously published LCA of UA. Multiple LCAs of the same farm or site were included when there were sufficient differences in production such as crop choice or use of inputs, since the LCA of that production system was deemed original. Defining whether a study met the requirements of an LCA was rather straightforward because it is a standardized method. We included papers that followed LCA methodology, meaning they evaluated several stages of production (not just on-farm activities) following the 'cradle to grave' principle of LCA, and modeled environmental impacts per unit output. LCAs typically consider multiple environmental impacts, but we included sources that only evaluated climate change ('carbon accounting') because of the dominance of this impact in the general LCA literature. Stated compliance with ISO 14040 was not a screening criterion, because we assumed that studies that did not meet the high standards of ISO were still useful in the context of this nascent research topic.

Defining whether a study evaluated UA was more difficult, and sometimes required informed decisions

based on context and descriptions of the case studies. This was necessary because there is no clear cut-off criterion for UA, and articles often did not provide complete descriptions justifying why the systems studied were UA. Although there is a commonly accepted definition of UA (based on its location in or around a town, city, or metropolis, and by its material and human links with the city), the application of this definition is dependent on the geographic and socio-economic urban landscape. This leads to substantial variation in what constitutes UA, including farm size, distance to the city center, and level of professional management. Generally, if an author defined the case study to be urban or peri-UA (PUA), we considered it UA. We erred on the side of inclusion, assuming that the expertise of authors led them to appropriately identify systems as UA. Due to important differences in farm size, level of professionalism, production methods, crop choices, and distribution pathways (Opitz *et al* 2016), we assessed results separately for intra-UA (IUA) and PUA.

We performed a second selection process for articles to include in the meta-analysis. Studies at the city level, and studies with uncommon functional units that could not be harmonized with impacts of food production at the farm level (for example, provisioning of resources needed to supply food, energy and water to a neighborhood for 1 year; or avoided impacts per kilogram of food produced), were excluded (such as Sanyé-Mengual *et al* 2017, 2018, Toboso-Chavero *et al* 2018, Weidner and Yang 2020). We also excluded consequential LCAs of UA, which is where only the consequences, or external changes resulting from a change in an activity, were modeled, rather than modeling the processes of a system itself (Benis and Ferrão 2017, Cleveland *et al* 2017, Puigdueta *et al* 2021). For example, modeling the reduced residential lawn maintenance if UA were to be installed, or the change in diets of UA practitioners. In two cases, we used the author theses (Perrin 2013, Fisher 2014) on which a publication was based (Fisher and Karunanithi 2014, Perrin *et al* 2015). One paper included a chicken production subsystem, which we excluded from the review due to the differences between animal agriculture and horticulture (Hall *et al* 2014).

2.2. Quantitative synthesis of the literature

In order to standardize functional units to kilograms of food produced per year and m^2 occupied per year, we performed basic conversions on the data provided regarding total food production, site area, days of operation, and cropping density. This is a common practice in meta-analyses of conventional agriculture (Poore and Nemecek 2018). These are shown in the supplementary material. When results were only available in figures, we used the software WebPlotDigitizer to extract the data.

Multiple production systems were often studied in each paper, so multiple sets of results were reported. This occurred when different farms were studied, or when different scenarios on one farm were evaluated. We recorded unique systems for the meta-analysis when there were substantially different production methods, such as different crops, substrates, lighting types (for indoor agriculture), or seasons. From the 38 papers evaluated for the meta-analysis, we identified 88 different farms or gardens, and 259 production systems.

## 3. Review results/synthesis

We present the descriptive results from the systematic literature review and then the results of our meta-analysis in three sections corresponding to our three research questions, followed by a section on the limitations.

3.1. Systematic review

3.1.1. Bibliometric trends

The majority of relevant literature studied cases in Europe (60% of papers), followed by Asia (20%), with scattered studies in North America (8%), South America (6%), Australia (4%) and Africa (2%) (figure 2). This global distribution of studies between economically developed and developing countries has been identified in other LCA reviews (Laurent *et al* 2014, Clune *et al* 2017, Poore and Nemecek 2018), and likely reflects the prevalence of LCA application rather than UA interest, which has been widely studied in developing regions (Zeeuw *et al* 2011, Orsini *et al* 2013). By country, most studies were done in Spain (31%), China (8%), Italy (8%), France (8%) and the United States (8%). By city, most studies took place in Barcelona, Spain (25%) and Beijing, China (8%). According to the Köppen climate classification, these cities are characterized by hot summers, and Barcelona has a main climate classification of warm temperate. Therefore, it may be important to note that many of these studies were done under favorable climatic conditions for agriculture. This is not evident for UA—particularly for controlled, indoor systems, which boast the potential to grow food in climates otherwise unfavorable to conventional agriculture.

We performed bibliographic network analyses with VOSviewer software and data from Scopus to identify clusters of researchers and prominent authors and publications (Van Eck and Waltman 2010). Figure 3(a) shows co-authorship through nearby placement of circles representing each author, which are more visible in figure 3(b), where we zoomed in on the main cluster. The literature is centralized around a cluster of researchers, where J Rieradevall was a co-author on ten papers (out of 47 total) and X Gabarrell and E Sanyé-Mengual each were co-authors in eight and seven papers. Many of these papers
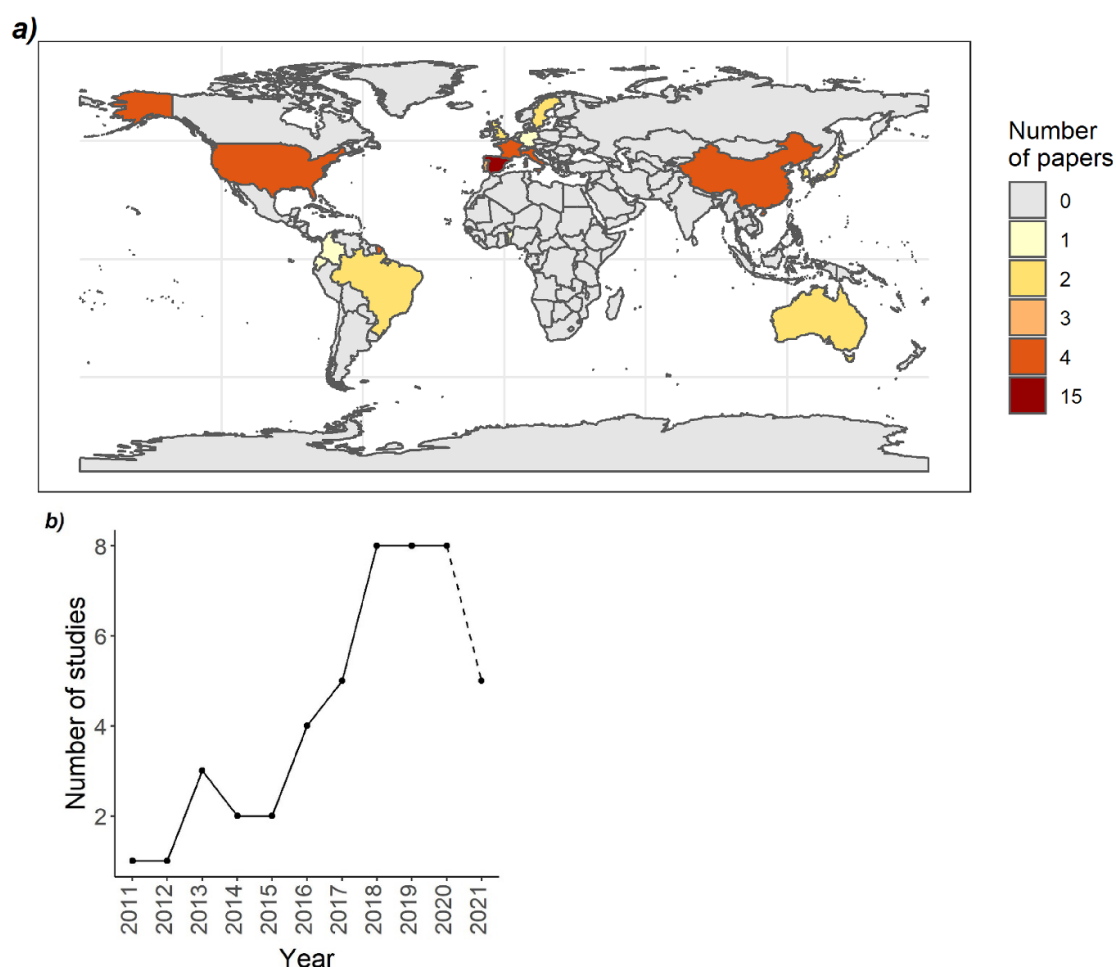
Figure 2. The map in part (a) shows that research performing LCAs of UA is largely centered in Europe, the United States, and China. Number of publications on the topic per year, shown in part (b), have increased over time, after first appearing a decade ago. This review only includes literature until April 2021, not the entire year, represented by the dashed line between 2020 and 2021.

focus on an experimental integrated RTG (iRTG) at the Universitat Autònoma de Barcelona in Spain, which 11 of the 47 papers evaluated, under different conditions (such as crop choice, growing methods, and implementation of rainwater harvesting system). In total, this cluster of researchers covers 37% of the papers, and the remaining were mostly isolated, meaning they were not co-authored with any other groups of authors from this set of literature. The top three cited papers were by Kulak *et al* (2013) with 104 citations, Sanyé-Mengual *et al* (2015a) with 84 citations, and He *et al* (2016) with 64 citations in Scopus as of April 2021.
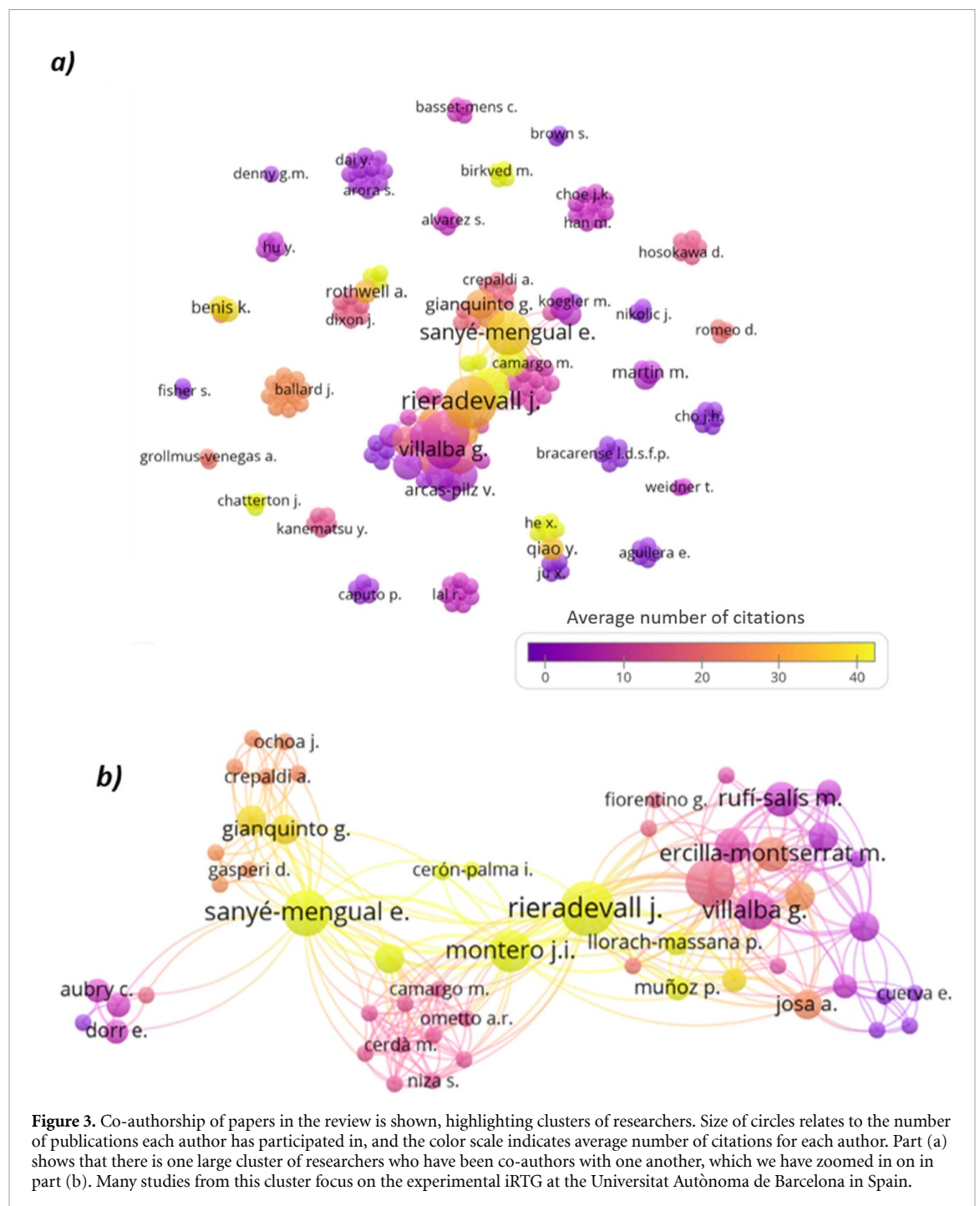
The Journal of Cleaner Production published the most papers on this topic, with about one third of the papers identified. The journal Sustainability followed, with five publications, and then Landscape and Urban Planning with three publications. Remaining journals had two or fewer publications each.

The first paper on the topic appeared in 2011, with a study of an indoor, vertical farm in Japan (Shiina *et al* 2011). This paper did not mention UA explicitly, so a study by Kulak *et al* (2013) is often identified as

the first LCA of UA. However, the study by Shiina *et al* was included in our review due to our inclusion of the keyword search for 'plant factory'. Publications per year generally increased since then, plateauing since 2018 at eight publications annually (figure 2(b)).

3.1.2. Framing and research objectives

The identified research was often framed in the context of UA being widely considered as less impactful than conventional agriculture, and aimed to test this paradigm (Fisher 2014, Sanyé-Mengual *et al* 2015b, Rothwell *et al* 2016, Goldstein *et al* 2016b, Romeo *et al* 2018), or to confirm and quantify the reduction in impacts from UA (Kulak *et al* 2013, Benis and Ferrão 2017, Cleveland *et al* 2017, Martinez *et al* 2018). Other work has pointed out a particular bias towards UA literature, focusing on and potentially overstating its benefits (Neilson and Rickards 2017, Weidner *et al* 2019). Another common framing is acknowledging that UA is becoming more and more prevalent, and even institutionally encouraged by policy makers, therefore it should become better understood and evaluated (He *et al* 2016, Ledesma

Figure 3. Co-authorship of papers in the review is shown, highlighting clusters of researchers. Size of circles relates to the number of publications each author has participated in, and the color scale indicates average number of citations for each author. Part (a) shows that there is one large cluster of researchers who have been co-authors with one another, which we have zoomed in on in part (b). Many studies from this cluster focus on the experimental iRTG at the Universitat Autònoma de Barcelona in Spain.

et al 2020). The literature generally shared the same objectives: to assess impacts of UA, identify hotspots in the life-cycle, and compare to conventional agriculture. Other common goals were to compare different forms of UA (Sanyé-Mengual *et al* 2015b, He *et al* 2016, Dorr *et al* 2017, Rufí-Salís *et al* 2020a) or identify ways to improve the management of specific UA systems (Liang *et al* 2019, Caputo *et al* 2020, Rufí-Salís *et al* 2020b).

3.1.3. Types of farms and gardens studied
The urban farms and gardens evaluated in the literature were highly diverse. Most papers studied IUA, but

eight papers evaluated PUA, covering 101 production systems. There is no commonly accepted typology for UA, so we categorized the farms among three important physical factors in order to aid our interpretation of the results: ground-based or rooftop, indoor or open-air, and hydroponics or soil-based. For this purpose, systems with growing media and technosols (soils created by human activity) were considered soil-based. PUA cases were mostly ground-based, open-air, soil-based production (91% of systems), but there were some ground-based, indoor, soil-based systems (9%) and one ground-based, open-air, hydroponics system. For IUA, papers generally

included only one of these physical forms, but 13% papers evaluated case studies from multiple types. The most frequently studied physical types for IUA were rooftop, indoor, hydroponics (32% of systems); ground-based, open-air, soil-based (25%); ground-based, indoor, hydroponics (22%); and rooftop, open-air, soil-based (11%). Indoor hydroponics systems are often described as vertical farms, RTGs or iRTGs.

Among the 39 papers studying IUA, 41% evaluated case studies with a research objective. Non-commercial systems (i.e. home gardens, school gardens, and non-profits) were the next most frequently studied, and were the focus of 18% of the papers. Commercial systems were represented in 15% of the papers. In 5% of the papers, multiple systems with different economic orientations were studied. For 21% of the papers, we could not define the economic orientation, and categorized those case studies as 'Unknown'. Among the eight PUA papers, 75% evaluated commercial case studies, and 12.5% were a home garden and 12.5% had an unknown economic orientation.

We found results for 45 different crops. Tomato and lettuce were the most frequently studied crops, appearing in 36% and 26% papers, followed by green bean (11%), arugula, basil, potato, and spinach (each in 6% of papers). The remaining crops were studied in only one or two papers. In 17% of the papers, LCA results were reported for a polyculture, or a mixture or 'basket' of crops. More than half of the papers only studied one crop (53%) or two (9%), and much of the diversity of crops studied came from a few papers mostly focusing on PUA, where 16–26 different crops were studied (Martinez *et al* 2018, Boneta *et al* 2019, Caputo *et al* 2020). We classified the crops into broad groups to simplify interpretation of the results, largely based on FAOSTAT categories (FAO 2020), although we sometimes adapted them to more appropriately show our data (for example we made tomatoes and leafy greens their own categories due to the large number of results) (table 1). Still, crops in the same category may have different crop cycle lengths, or growing requirements, so results are also shown per crop in figures and in the supplementary material. The most frequently studied crop categories were tomatoes, leafy greens, and then vegetables. Together, these groups appeared in 79% of papers studying IUA, and 38% for PUA. Cereals and legumes were infrequently studied, which was not surprising, because these crops are generally not cultivated in UA.

Data about the size of farms were available for about 75% of the systems, and there were important differences between IUA and PUA. For IUA, total farm area ranged from 18 to 32 728 m^2, with a mean and median of 1608 and 245 m^2. The average farm area in cultivation was 268 m^2, and the average percent of the farm area in cultivation was 69%. Within IUA, ground-based systems were usually larger than

Table 1. The 45 different crops evaluated in the literature were broken down into categories, largely based on the categorization in FAOSTAT. Many results through this review are reported by crop category rather than specific crop, but results per crop can be found in the supplementary material.

Crop category	Crops included
Tomato	Tomato, cherry tomato
Leafy greens	Arugula, cabbage, chard, chicory, lettuce, leafy greens, pak choi, spinach
Herbs	Basil
Fruits	Apple, cherry, fig, melon, mixed berries, mulberry, peach, plum, pomegranate, sorb, strawberry, watermelon
Vegetables	Asparagus, artichoke, bell pepper, chili pepper, eggplant, green bean, kohlrabi, mushroom, pumpkin, zucchini
Cereals	Barley, maize, millet, spelt, wheat
Legumes	Chickpeas, lentils
Polyculture	Polyculture
Roots and tubers	Carrot, onion, potato, radish, sweet potato

rooftop systems, with average cultivated areas of 348 and 225 m^2, respectively. IUA systems with the largest cultivated area were found in Asia (average of 783 m^2) followed by North America (average 634 m^2), and the smallest farms were found in Europe (average of 135 m^2). For PUA, the mean and median of total farm area were 38 881 and 22 000 m^2, and the average area in cultivation was 15 308 m^2. On average, the percent area in cultivation for PUA was 55%.

It is impossible to evaluate whether these studies are representative of UA because there is no global survey on the nature of UA. However, surveys of specific types of UA in specific geographic boundaries can give an indication. Appolloni *et al* (2021) surveyed global rooftop UA and found that most are open-air (84%), suggesting that the large number of indoor systems (66% of rooftop systems) studied with LCA may be outsized. They found that a majority of rooftop UA was not commercial, and had primarily socio-economic goals, but in this review most rooftop systems were oriented towards research. Numerous studies indicate that lettuce and tomato are the most prominent crops grown across various forms of UA, followed by beans and herbs (Pourias *et al* 2015, Buehler and Junge 2016, Appolloni *et al* 2021). This crop choice is well reflected in the literature in this review.

3.1.4. Data collection and system modeling
A number of different methods were used by researchers to collect data from UA systems, including directly measuring data, consulting operations records, interviewing farmers and gardeners, distributing surveys to farmers and gardeners,

modeling relevant scenarios based on limited data, and using values found in the literature (from UA and conventional agriculture). About half of the papers (49%) used only mass of food produced as the functional unit (kilogram or ton of food produced per year). This may be problematic for UA where the main function is not always to produce food, but rather is multifunctional, and may not perform best according to its food production objectives. Also, limiting the functional unit to mass neglects other functions of agriculture (such as land stewardship) and food (provide nutrition, protein, food quality), and lightweight crops like herbs are inherently penalized when comparing to water-heavy crops like tomato. A parallel example is organic agriculture, which usually performs worse than conventional when using a food mass-based functional unit (impacts per kilogram or ton food produced per year), due to lower yields. However, using an area-based functional unit (impacts per m^2 or hectare cultivated per year), organic agriculture consistently performs better than conventional agriculture (Meier *et al* 2015). Most of the time the functional unit was kilogram of a specific crop grown, and in 16% of papers, kilograms of mixed crops or a polyculture were used. After food production, the most common functional unit was land use (m^2 or hectare per year), which appeared in 20% of papers. Other functional units included annual or lifetime operations at a farm/garden, annual food consumption needs of inhabitants, calories produced, and revenue.

In 26% of the papers we reviewed, authors stated that there was no transport necessary to the consumer, because, for example, the consumer lived in the building on which food was produced, or consumers were situated very close and walked to the farm/garden. For the purpose of this review, we classified the system boundary in these cases as including post-farm delivery, because the farm-to-consumer stage was considered, even though there were no processes or impacts. Given this re-classification, 70% of the papers used a system boundary that accounted for distribution to the consumer or to retail (with or without actual transport processes), and 30% considered processes only up to the farm level.

About half of the studies used the software SimaPro to perform the LCAs. In 40% of the papers, no specific software was mentioned. The remaining studies used OpenLCA, TRACI, or Excel.

Four papers modeled the impacts of off-farm changes as a result of implementing UA. These studies used a 'consequential' system modeling approach for LCA, where the consequences of a change to a system are modeled rather than the processes of an actual system (which is an 'attributional' approach). Benis *et al* (2017) modeled the hypothetical reduced transport distances of produce, reduced waste along the supply chain, and shifts towards the recommended healthy diet that may come with UA in Lisbon, Portugal.

They found that diet changes (notably from reduced meat consumption) contributed the most to reducing climate change and land use impacts. Cleveland *et al* (2017) similarly modeled hypothetical changes associated with implementing UA in Santa Barbara, California, but focused on household gardens and different outcomes: reduced impacts from less lawn maintenance, avoided conventional vegetable purchases, and reduced municipal organic waste and wastewater treatment; and increased impacts from composting emissions at home gardens. They found that avoided municipal organic waste treatment brought the largest reductions in climate change impacts. Oliveira *et al* (2021) used survey responses to model the estimated changes in distribution logistics for lettuce produced in UA in Belo Horizonte, Brazil, considering reductions in municipal organic waste collection, reduced transport steps for distribution, and changes in transport modes, and found that the UA system had lower climate change and human toxicity impacts by 76% and 67%, respectively, compared to the current system. Puigdueta *et al* (2021) used responses from a longitudinal (5 year) survey on food consumption patterns in Madrid, Spain, from a group of novice community gardeners and a control group not involved in UA. They modeled changes in organic food consumption, the shift to a 'low-carbon' diet, waste generation and treatment practices, and change in transport patterns for food purchases, among other factors. They found that changes in diet linked to social learning at gardens drove the largest reduction in climate change impacts (especially reduced meat consumption). The reduced climate change impacts in the UA group were 9% larger than the reductions in the control group. Such studies are relevant to explore the importance of these benefits, but can be misleading because they do not include the processes and impacts of actually operating UA. Interpretations that have been drawn from these results as full life-cycle based evidence of large climate change reductions by UA are misguided because only the reductions were modeled, and not the actual full impacts of UA (Cleveland *et al* 2017, Vávra *et al* 2018).

3.2. Summary of the environmental performance and resource efficiency of UA

Some measures that do not have a life-cycle perspective but are nonetheless useful environmental indicators are yield (food produced for a given area in a year) and water use efficiency (amount of water used for growing a given amount of food). These represent efficient use of land and water, which are typically dominant at the farm stage of a food LCA. We summarized these non-life-cycle indicators in the meta-analysis as well. We focused on tomato and lettuce because they were the most frequently studied crops (although we present results from each crop grown in figures and in the supplementary material). This reflects the fact that tomato and lettuce are commonly

cultivated in UA, and they are important in the diets where these LCAs were done.

The summary of life-cycle impacts was restricted to only climate change and cumulative energy demand (CED), because these impact categories can be reliably compared across papers even when different impact assessment methods are used. Other indicators were measured with a variety of impact assessment methods, such as ReCiPe and ILCD (European Commission, Joint Research Centre 2011, Huijbregts *et al* 2017), which can be based on fundamentally different environmental fate and impact models. We summarized many of these results, and the impact assessment methods used in the supplementary material, for future comparisons.

Generally, climate change impacts for UA can be expected between 0.03 and 4 kg CO_2 eq. kg^{-1} crop harvested per year (75% of results fall within this range). Results by crop are provided in detail below. For indoor IUA systems, there was a wider spread and larger (sometimes much larger) impacts could be found, but still two thirds of the results were less than 4 kg CO_2 eq. kg^{-1} crop. Open-air IUA systems, in contrast, had a narrower distribution and smaller climate change impacts (75% of results were less than 2.1 kg CO_2 eq. kg^{-1} crop). Based on area, climate change impacts for IUA can be expected between 0.5 and 20 kg CO_2 eq. m^{-2}, and should still generally be higher in indoor than open-air systems. For PUA, impacts were smaller and there was less variation. CED was usually less than 10 kWh kg^{-1} crop, and less than 2 kWh kg^{-1} crop for non-renewable CED (36 and 7 MJ kg^{-1} crop, respectively). Yields for open-air UA were usually between 0.5 and 4.75 kg fresh weight crop harvested m^{-2} (75% of values) and for indoor UA was more spread out, with about two-thirds of the values below 34 kg m^{-2}. Water use varied widely and was mostly between 0.2 and 150 l kg^{-1} crop (75% of values).

3.2.1. Yield

We found yields for 77% of the production systems. Yields varied widely, with a mean of 16 ± 33 kg m^{-2} and median of 2.4 kg m^{-2} (both in fresh weight). These values represent total harvest, and losses on the farm or in distribution were either not mentioned in the literature, or authors specified that there were no losses. By crop category, the highest average yields were found for herbs, followed by leafy greens (using the mean only) and tomato (which had a median higher than leafy greens) (figure 4(a)). This was likely because these crops were frequently grown in indoor, vertical farms. Polycultures, roots and tubers, and vegetables had the next highest yields. Fruits, grains, and legumes, which were only reported for PUA, had the lowest yields, which may reflect the open-air, soil-based systems where they are typically cultivated. Tomato and lettuce, the most frequently studied

crops, had average overall yields of 15 ± 16 kg m^{-2} and 17 ± 33 kg m^{-2}, respectively. In open-air, soil-based systems, tomato and lettuce had average yields of 6.4 ± 5.5 kg m^{-2} and 2.6 ± 1.5 kg m^{-2}. A breakdown of yields for each crop in different physical farm types is in the supplementary material. By production system type, for IUA, the highest yields were found in ground-based indoor hydroponics systems, followed by ground-based indoor soil-based systems (figure 4(b)). Rooftop indoor hydroponics systems had a large mean yield (7.6 kg m^{-2}) but a small median (0.59 kg m^{-2}), because one farm with many systems grew a variety of vegetables in a research setting with rather low plant densities (Rufí-Salís *et al* 2020a, 2020b, Arcas-Pilz *et al* 2021). For this physical farm type, there was a clear distinction between crops, where tomato yields (21 ± 17 kg m^{-2}) were much larger than lettuce (3.2 ± 4.7 kg m^{-2}) and vegetable yields (0.40 ± 0.25 kg m^{-2}, mostly green beans). All types of open-air systems had lower yields than indoor systems, and soil-based systems had larger yields than hydroponics in open-air. The distribution of yields was skewed to the right, with many smaller yields reported (2/3 of the values below 6 kg m^{-2}, which is actually relatively good, as shown in the next paragraph) and a handful of very large yields. Systems with the largest yields, over 100 kg m^{-2}, came from several different papers, and were 'vertical farms' or 'plant factories' with artificial lighting, temperature control, and strategic use of the vertical dimension with stacked floors of crop production (Shiina *et al* 2011, Martin and Molin 2019, Martin *et al* 2019, Pennisi *et al* 2019). There was not a clear distribution in yields across different climates for open-air systems. Average yields were much higher in commercial systems than in non-commercial systems (figure 4(c)), which is likely due to a combination of factors including farm management and the physical set-up, where indoor systems were more often found in commercial endeavors.

For reference, we compared these values to averages from FAOSTAT, over the most recent 5 year period available (from 2014 to 2018), for countries/regions that were commonly studied in the literature: the European Union, Spain, the United States, and mainland China (FAO 2020). For tomato, yields ranged from 5.0 to 9.3 kg m^{-2}, with an average of 7.1 kg m^{-2}. In the UA systems, open-air, soil-based tomatoes had similar yields, and the average yield including all production sites was more than twice as large. For lettuce, the FAOSTAT yields ranged from 2.4 to 3.5 kg m^{-2}, with an average of 2.7 kg m^{-2}. As with tomato, this was similar to open-air soil-based UA yields, and the overall UA yields were much higher thanks to indoor hydroponics systems. A grouped category of 20 vegetables and greens from FAOSTAT showed an average yield of 3.1 kg m^{-2} over the selected years and locations, which is lower than the UA average yield for all open-air, soil-based production.
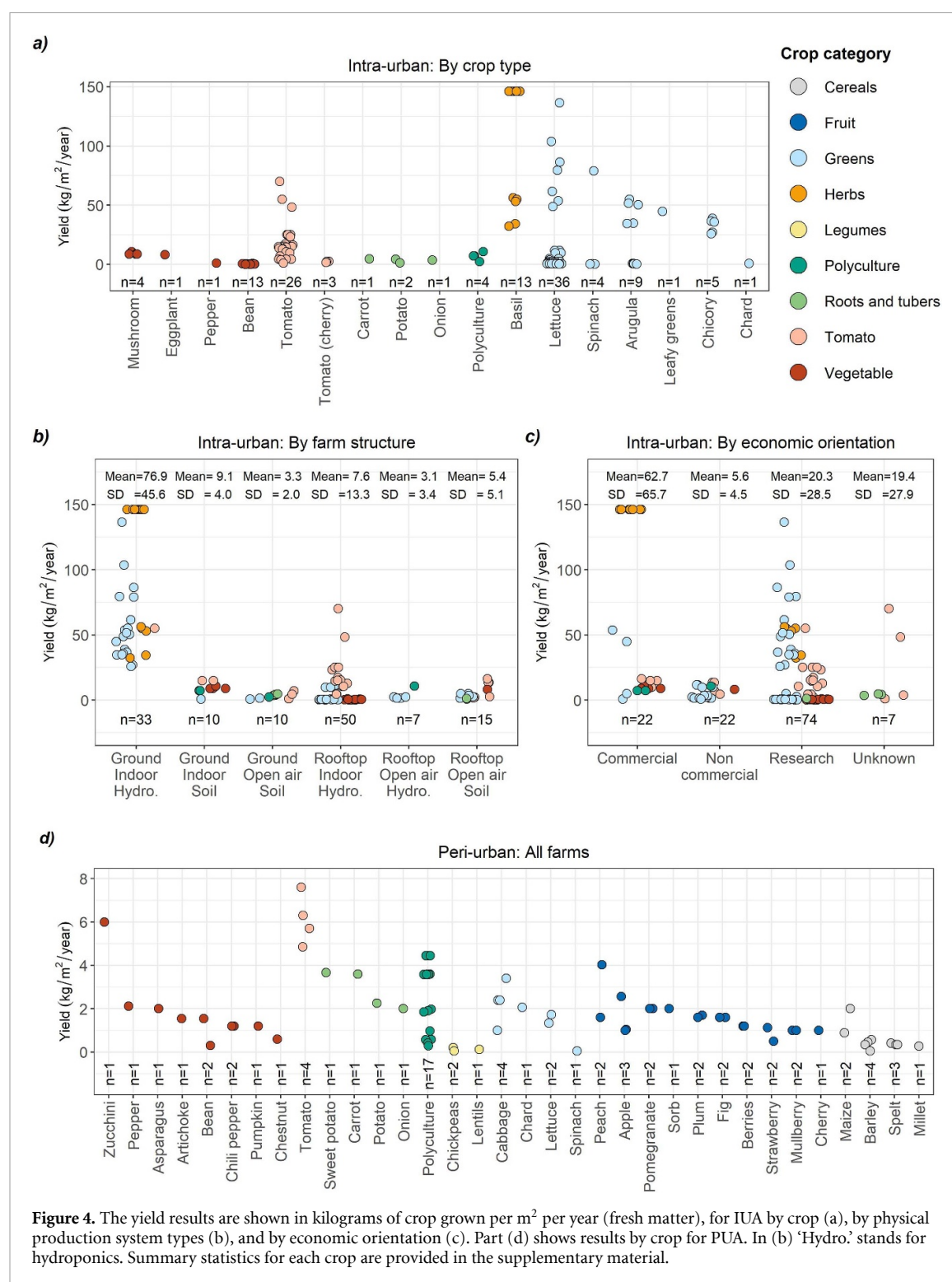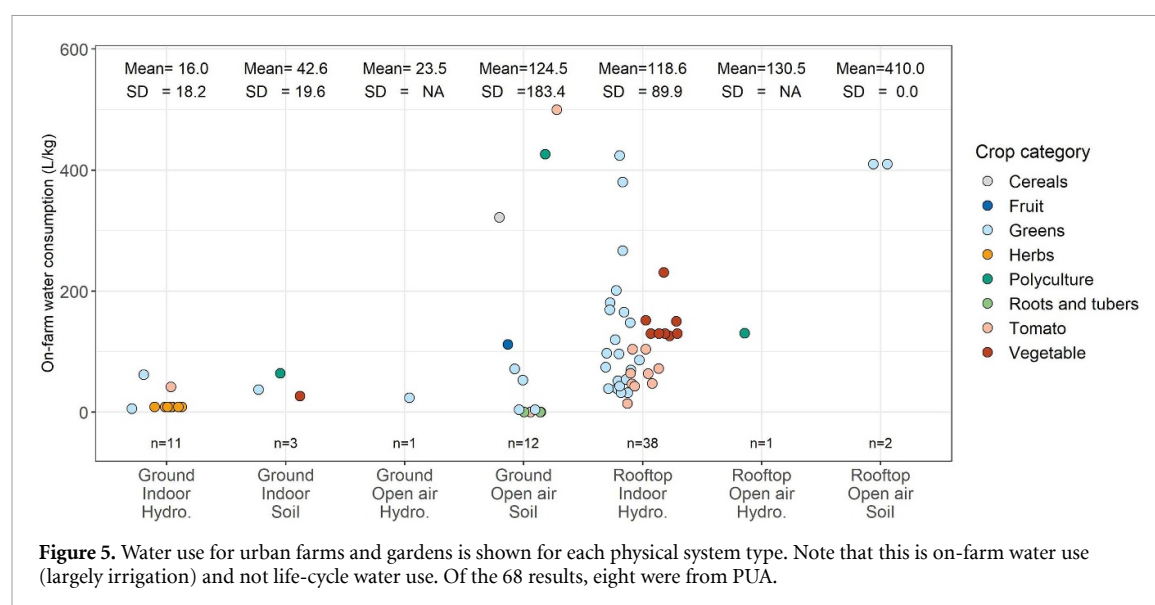
Figure 4. The yield results are shown in kilograms of crop grown per m² per year (fresh matter), for IUA by crop (a), by physical production system types (b), and by economic orientation (c). Part (d) shows results by crop for PUA. In (b) 'Hydro.' stands for hydroponics. Summary statistics for each crop are provided in the supplementary material.

A review by Poore and Nemecek (2018) found similar average yields for conventional agriculture, in the range of 2.5–4 kg m⁻², for tomatoes, onions, leeks, root vegetables, and brassicas. These are imprecise comparisons, but nonetheless encourage that the UA systems studied had yields that were at least on par with, and sometimes much greater than, conventional agriculture.

3.2.2. Water use

On-farm water consumption data were available for 68 production systems from 16 different papers. This represents blue water consumption from irrigation, and does not account for green water consumption from rainfall. The liters used per kilogram of food produced ranged from 0.16 to 500 l kg⁻¹, with a mean of 107 ± 121 l kg⁻¹. Water consumption

Figure 5. Water use for urban farms and gardens is shown for each physical system type. Note that this is on-farm water use (largely irrigation) and not life-cycle water use. Of the 68 results, eight were from PUA.

was similar for IUA (103 ± 117 l kg^{-1}) and PUA (139 ± 150 l kg^{-1}). The average water consumption for lettuce and tomato was 93 ± 106 and 92 ± 132 l kg^{-1}, respectively. This was measured for all types of systems except for ground-based open-air hydroponics (figure 5). Spinach and beans had larger water consumption, with 357 ± 81 and 150 ± 37 l kg^{-1}, respectively, measured only in indoor hydroponics systems. In comparison, global averages of blue water footprints from the years 1996–2005 were 66 l kg^{-1} for tomato, 28 l kg^{-1} for lettuce, 54 l kg^{-1} for green beans, and 14 l kg^{-1} for spinach (Mekonnen and Hoekstra 2010). Additionally, a review of conventional vegetable LCAs found that, among 72 systems, 80% had irrigation amounts below 100 l kg^{-1}, compared to 60% for the UA systems here (Perrin *et al* 2014).

Rooftop open-air soil-based systems had the largest average water consumption, although there were only two results of for this type (figure 5). Overall, the results were particularly skewed by a few large measurements: seven records with water consumption greater than 300 l kg^{-1}. These extreme records came from four different papers, and diverse production systems (both open-air soil and rooftop indoor hydroponics), crop types (tomato, greens, maize, and a polyculture), both IUA and PUA, and global regions with different climates (Barcelona, Spain; Quito, Ecuador; Beijing, China; multiple cities in Benin), suggesting that extreme water consumption may be an uncommon but possible facet of UA (Perrin 2013, Liang *et al* 2019, Ledesma *et al* 2020, Rufí-Salís *et al* 2020a).
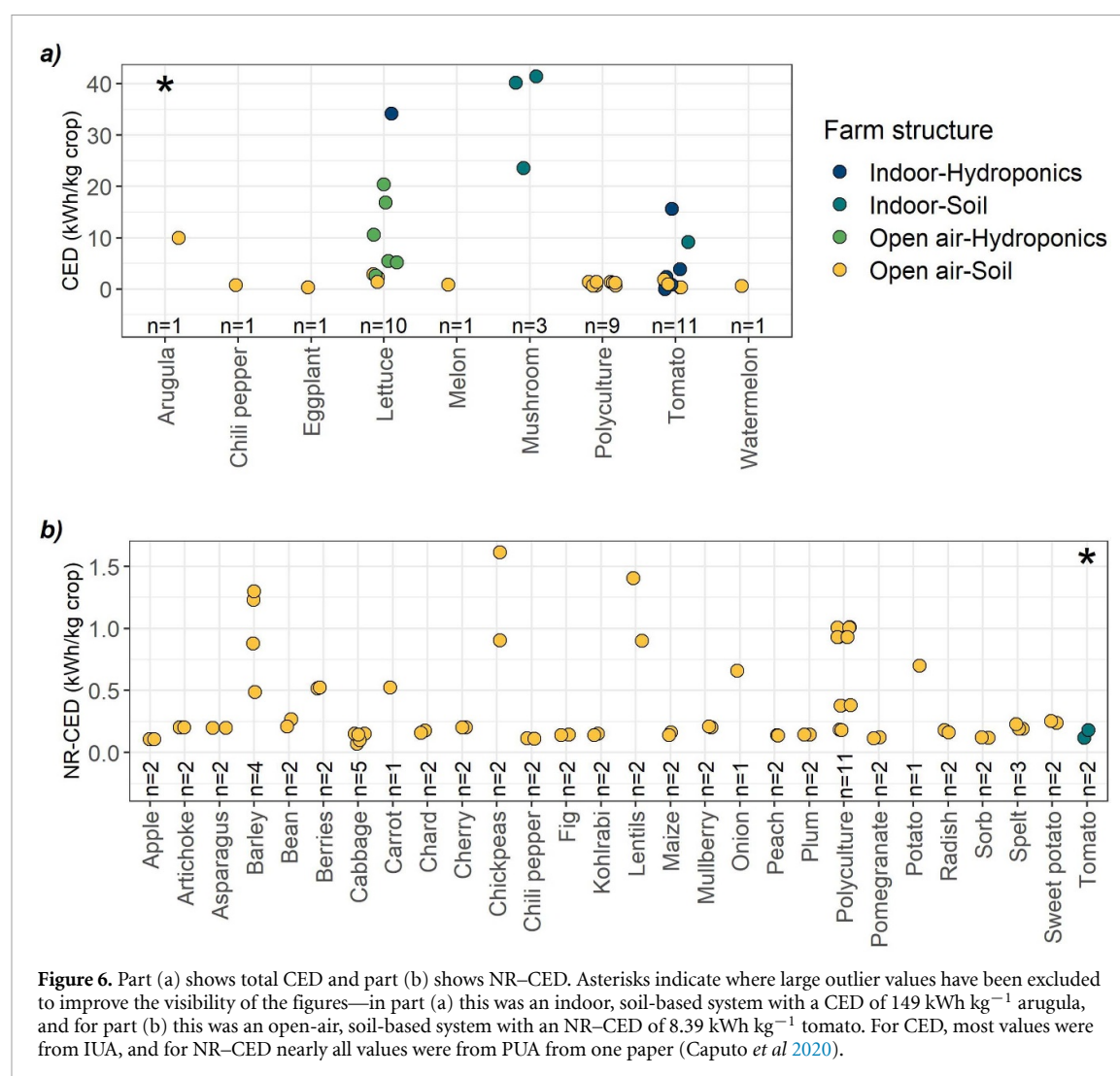
3.2.3. Cumulative energy demand results
There were results from 69 production systems for non-renewable CED (NR–CED) and 39 for total CED, from four and seven different papers, respectively (figure 6). Most CED results were from

IUA systems (75%), and nearly all NR–CED results were from PUA (94%), specifically from one paper (Caputo *et al* 2020). Among the CED results for IUA, rooftop, open-air, soil-based systems had the lowest impacts, with a mean and median of 0.94 and 0.78 kWh kg^{-1} crop, followed by ground-based, open-air, soil-based systems with a mean and median of 3.7 and 4.2 kWh kg^{-1} crop. Rooftop indoor hydroponics systems had the next largest CED (mean and median of 4.5 and 2.3 kWh kg^{-1} crop), and ground-based, indoor, soil-based systems had the largest CED (mean and median of 53 and 40 kWh kg^{-1} crop). We found multiple CED results for the following crops: 79 and 149 kWh kg^{-1} arugula in indoor and open-air soil-based systems, 35 ± 9.9 kWh kg^{-1} mushroom in indoor systems, 10 ± 11 kWh kg^{-1} lettuce in nearly all different physical setups, and 3.3 ± 4.8 kWh kg^{-1} tomato also from various system types. For PUA, the mean and standard deviation of NR–CED and CED were 0.37 ± 0.38 kWh kg^{-1} crop and 1.08 ± 0.32 kWh kg^{-1} crop. We found positive correlations between climate change impacts per kilogram of crop and both NR–CED ($r = 0.95$, p-value $= 2.2 \times 10^{-16}$) and CED ($r = 0.96$, p-value $= 2.2 \times 10^{-16}$). In most cases, we were not able to distinguish between direct and indirect energy use for these systems. Still, several results from an indoor mushroom farm, extensive peri-urban farm, and RTG showed direct, on-farm energy use contributed 66%, 48%, and 38%–53% of CED, respectively. The remaining energy use came from distribution and embodied energy use.

3.2.4. Climate change
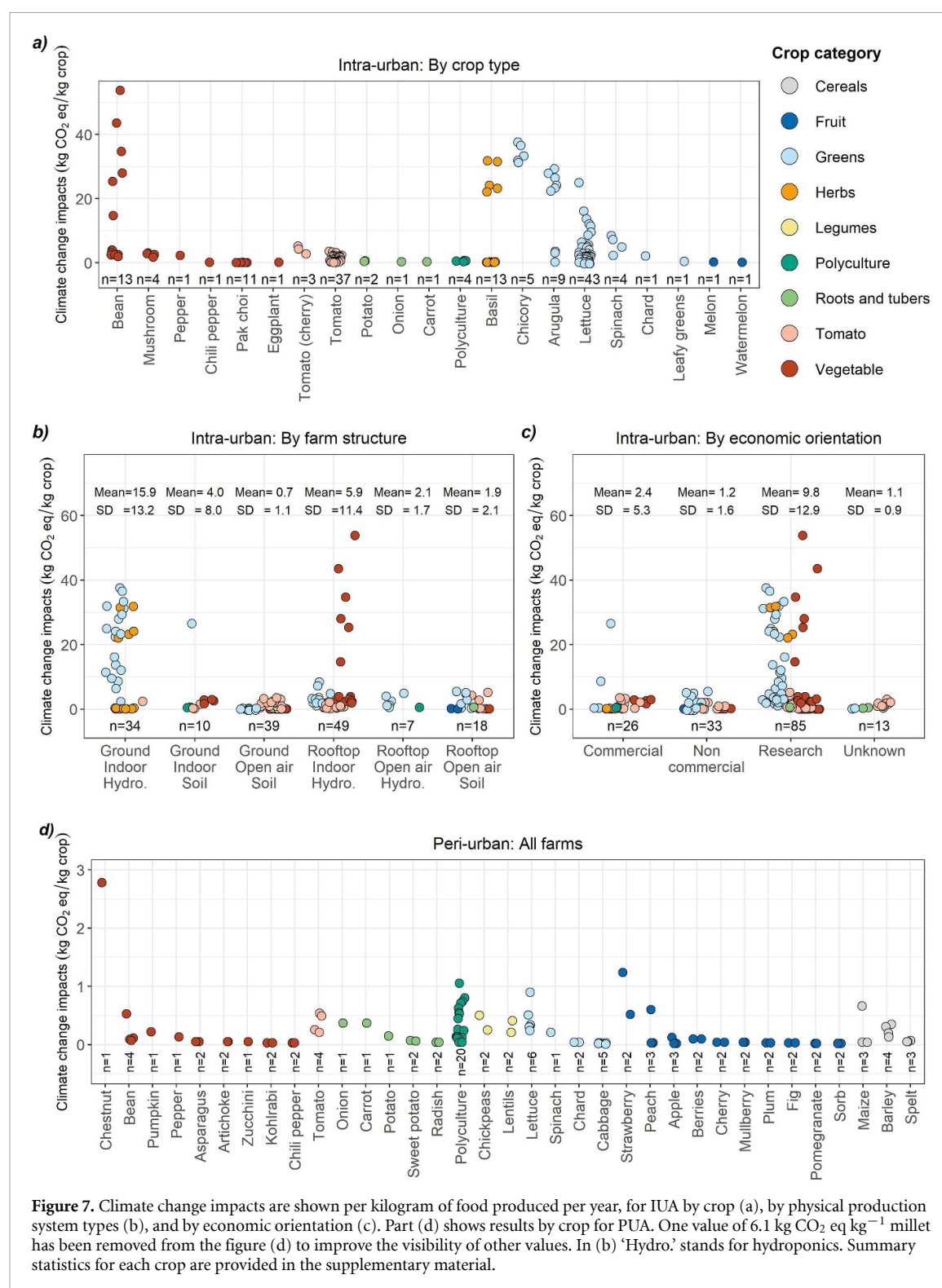The climate change results per kilogram of crop differed by a factor of 5000, with positive values ranging from 0.01 to 54 kg CO_2 eq. kg^{-1} crop (negative emissions were even sometimes found due to avoided products, described below). The mean for

Figure 6. Part (a) shows total CED and part (b) shows NR–CED. Asterisks indicate where large outlier values have been excluded to improve the visibility of the figures—in part (a) this was an indoor, soil-based system with a CED of 149 kWh kg^{-1} arugula, and for part (b) this was an open-air, soil-based system with an NR–CED of 8.39 kWh kg^{-1} tomato. For CED, most values were from IUA, and for NR–CED nearly all values were from PUA from one paper (Caputo *et al* 2020).

IUA systems was 6.0 ± 11 kg CO$_2$ eq. kg^{-1} crop, and the median was 1.83 kg CO$_2$ eq. kg^{-1} crop. The breakdown of impacts by crop are shown in figure 7(a), and statistical summaries for each crop are in the supplementary material. The most frequently studied crops, tomato and lettuce, had average impacts in IUA of 1.4 ± 1.2 kg CO$_2$ eq. kg^{-1} tomato and 4.2 ± 5.2 kg CO$_2$ eq. kg^{-1} lettuce. Between IUA systems, ground-based, indoor, hydroponics had the largest impacts (figure 7(b)). The second largest impacts came from rooftop, indoor, hydroponics systems, where 88% of the results came from the same RTG at the Universitat Autònoma de Barcelona in Spain. Ground-based, open-air, soil-based systems, which are most similar to a conventional agriculture setup, had the lowest impacts. As with yield, the rooftop–ground-based dimension was especially relevant for indoor hydroponics systems, where ground-based ones (often called 'vertical farms') had larger impacts than rooftop ones (RTGs), despite their increased efficiency in growing food, evidenced by higher yields.

Non-commercial IUA systems had lower impacts than commercial ones when looking at the mean, but one large outlier value of 27 kg CO$_2$ eq. kg^{-1} crop skewed the mean of commercial systems. Looking at the median, commercial systems had lower impacts than non-commercial ones (0.44 and 0.55 kg CO$_2$ eq. kg^{-1} crop, respectively). Systems used primarily for research had the largest impacts (figure 7(c)). Numerous systems used experimental production methods, including using biochar and struvite as inputs (Shen *et al* 2020, Arcas-Pilz *et al* 2021), recirculating nutrients in hydroponics systems (Rufí-Salís *et al* 2020b), testing different LED lighting schemes (Pennisi *et al* 2019), or using waste such as spent coffee grounds and brewers' grains for substrates (Martin *et al* 2019, Dorr *et al* 2021), which led to reduced yields, and may not be representative of how such systems would perform after research leads to improvements. Similarly, the iRTG in Barcelona, which was one of the first of its kind and the source of many results in this review, contributed large climate change impacts from its infrastructure, but numerous improvements have been identified that would reduce impacts in future systems (Muñoz-Liesa *et al* 2021). It seems that numerous results here do not reflect a snapshot of current UA, but rather show the sub-optimized first

Figure 7. Climate change impacts are shown per kilogram of food produced per year, for IUA by crop (a), by physical production system types (b), and by economic orientation (c). Part (d) shows results by crop for PUA. One value of 6.1 kg CO_2 eq kg^{-1} millet has been removed from the figure (d) to improve the visibility of other values. In (b) 'Hydro.' stands for hydroponics. Summary statistics for each crop are provided in the supplementary material.

iterations of potential production methods for the future.

The distribution of climate change impacts for IUA show that the results were skewed by a handful of systems with particularly high impacts, as was found in the review of thousands of food products by Poore and Nemecek (2018). The 39 IUA systems (out of 157 total IUA systems) with impacts above the 75th percentile (4.0 kg CO_2 eq. kg^{-1} crop) came from

nine different papers, five different physical setup types, and seven different crops, suggesting that they were not anomalies attributed to inconsistent modeling choices or unique systems. A similar skew was found for the yield results. Many of the largest climate change impacts came from Pennisi *et al* (2019), where 19 systems had impacts greater than 10 kg CO_2 eq. kg^{-1} of greens or herbs (with a mean of 25 kg CO_2 eq.). This was a small-scale experimental setup at the

University of Bologna comparing the effects of different ratios of red and blue light in hydroponics systems, in small compartments of 0.6 m^2, and also had among the highest yields, CED, and area-based climate change impacts. Most impacts came from electricity use for lighting, and the authors noted that the experimental prototype lamps used were less efficient (in terms of μmol Joule^{-1}) than commercial versions of the same lamps. If we exclude results from this paper, the mean climate change impacts for the remaining 14 indoor, hydroponics, ground-based systems is 3.33 $\pm$ 6.8 kg CO$_2$ eq. kg^{-1} crop, which is comparable to other systems. Other large impacts came from Arcas-Pilz *et al* (2021), where the six hydroponics systems studied at the iRTG in Barcelona had impacts between 14.7 and 53.8 kg CO$_2$ eq. kg^{-1} of green bean. They compared four systems with varying amounts of struvite fertilizer (for phosphorus) and rhizobium inoculation (for nitrogen) to two control systems with mineral fertilizer inputs. They found that infrastructure (the greenhouse structure and rainwater harvesting system) accounted for more than 90% and 75% of impacts in struvite and control systems, and all systems had very low yields (0.07–0.29 kg crop m^{-2}), partly due to short cropping periods (only 84 d, which was temporally accounted for in the allocation of infrastructure) and growing in winter and early spring. Here it seems that the environmentally heavy fixed impacts of infrastructure were not compensated by similarly high yields, even when accounting for the short period of time the infrastructure is used for (which is a near-universal practice in LCA). One system in Goldstein *et al* (2016b) had climate change impacts of 26.5 kg CO$_2$ eq. kg^{-1} of arugula, due to large heating demands of a greenhouse in Boston, USA (the CED was 149 kWh kg^{-1}, which was also quite large), combined with low yields of 0.7 kg crop m^{-2}.

On the other end of the spectrum, extremely low results were found across many studies: seven papers had systems with less than 0.1 kg CO$_2$ eq. kg^{-1} crop (five from IUA, two from PUA). These were generally the result of systems with few inputs thanks to environmentally inert materials (like reused materials), simple production systems (for example with no irrigation, infrastructure, fertilizer, or compost), limited LCAs excluding some processes, or systems with no need for (or excluded) distribution. One study that found very low on-farm impacts for pak choi (0.03–0.11 kg CO$_2$ eq. kg^{-1} crop, based on data for soil-based lettuce production from the Ecoinvent database) found that when accounting for the external benefits from using biochar (avoided wood waste incineration), impacts were further reduced to -0.02 to -19 kg CO$_2$ eq. kg^{-1} crop (Shen *et al* 2020). It can be difficult to interpret the real meaning of negative impacts from LCAs, but we can consider that these systems not only have small impacts compared to similar systems, but actually reduce greenhouse gas emissions by sequestering carbon or causing impactful processes to be avoided.

Climate change impacts were also evaluated based on the land-use function of UA, using a functional unit of 1 m^2 of land occupied for 1 year (figure 8). This was provided directly for 81 systems, and we were able to calculate it for 98 additional systems using basic conversions with yield data. Considering the land occupation function rather than food production is a useful way to focus on the farm and garden operations regardless of efficient food production, which is not a focus of many UA projects. The mean, standard deviation, and median for IUA were 79 $\pm$ 237 kg CO$_2$ eq. m^{-2} and 4.7 kg CO$_2$ eq. m^{-2}. Results were largely influenced by 12 systems from four different papers with large values of 131–986 kg CO$_2$ eq. m^{-2}, from ten leafy greens and two tomato systems, in ground-based, indoor, hydroponics systems (but one rooftop), mostly research systems (and one commercial and one with unknown economic orientation) (Shiina *et al* 2011, Goldstein *et al* 2016b, Kikuchi *et al* 2018, Pennisi *et al* 2019). These systems also had large average yields (70 $\pm$ 30 kg m^{-2}) and food-based climate change impacts (12 $\pm$ 10 kg CO$_2$ eq. kg^{-1} crop). Energy use for lighting and temperature regulation are the top contributors to climate change impacts in all of these systems. These systems are not shown in figure 8 to improve readability, but they are included in the calculations of mean, standard deviation, and number of observations. Similar trends were seen as in the climate change impacts based on food production, where greens, herbs, research, and indoor hydroponics systems had the largest impacts. However, by area, the relative impact of ground-based indoor hydroponics was more exaggerated, partially due to the very large values described above, but also due to larger values within the 'normal' range of results. This could be expected because when evaluating by area, we do not account for the compensation of large inputs with large yields. Similar to the food-based impacts, PUA had lower impacts than IUA, with a mean of 0.51 $\pm$ 0.90 kg CO$_2$ eq. m^{-2} and 0.14 kg CO$_2$ eq. m^{-2}.

3.2.4.1. Comparing climate change impacts of UA to conventional agriculture

Regardless of whether or not UA is positioned to compete with conventional agriculture, we often compare their environmental impacts, which at least provides a frame of reference. At the same time, there are numerous examples of UA systems positioned to compete with conventional agriculture, and in those instances such comparisons are more justified. Our first method of comparing UA climate change impacts to conventional agriculture was using the in-paper, pair-wise comparisons. About half of the UA production systems were compared by authors to specific, local, conventional agriculture systems of the same crop. Climate change impacts per kilogram of
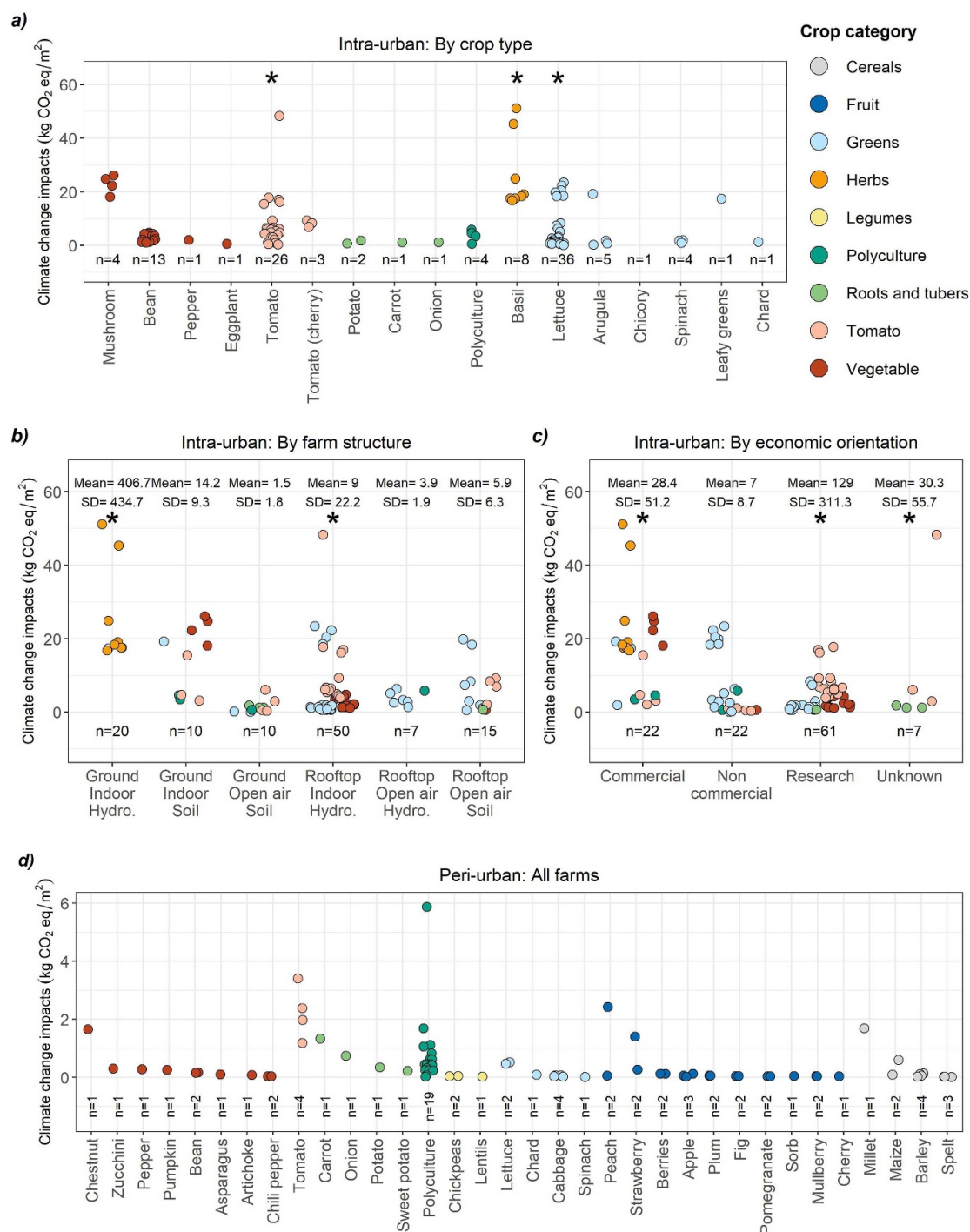
Figure 8. Climate change impacts per m² land occupied per year are summarized here, for IUA by crop (a), by physical production system types (b), and by economic orientation (c). Part (d) shows results by crop for PUA. In (b) 'Hydro.' stands for hydroponics. Note that 12 outlier points have been excluded, with values of 130–985 kg CO_2 eq m^{-2}, from mostly leafy greens and two tomato systems, in indoor, hydroponics, rooftop (one ground based), research systems (and one commercial and one with unknown economic orientation), from four different papers. These values have, however, been included in calculation of the mean, standard deviation and number of observations. Asterisks show the groups where values have been excluded.

crop from IUA were lower than the conventional system in 41 out of 68 comparisons (60%), and higher in 40% of comparisons (figure 9). In almost all PUA systems, climate change impacts were lower than for conventional agriculture (96%). Indoor hydroponics systems and leafy green crops generally performed worst against their conventional agriculture comparison (figure 9(a)). Open-air, soil-based IUA systems

and IUA tomatoes usually performed better than conventional agriculture (figures 9(b)–(d)).

Our second method of comparing UA impacts to conventional agriculture was using generalized results from food and agriculture LCA reviews (figure 10). Although these comparisons may be less precise, with different climate and local contexts, the large sample size made them more representative. The outcomes
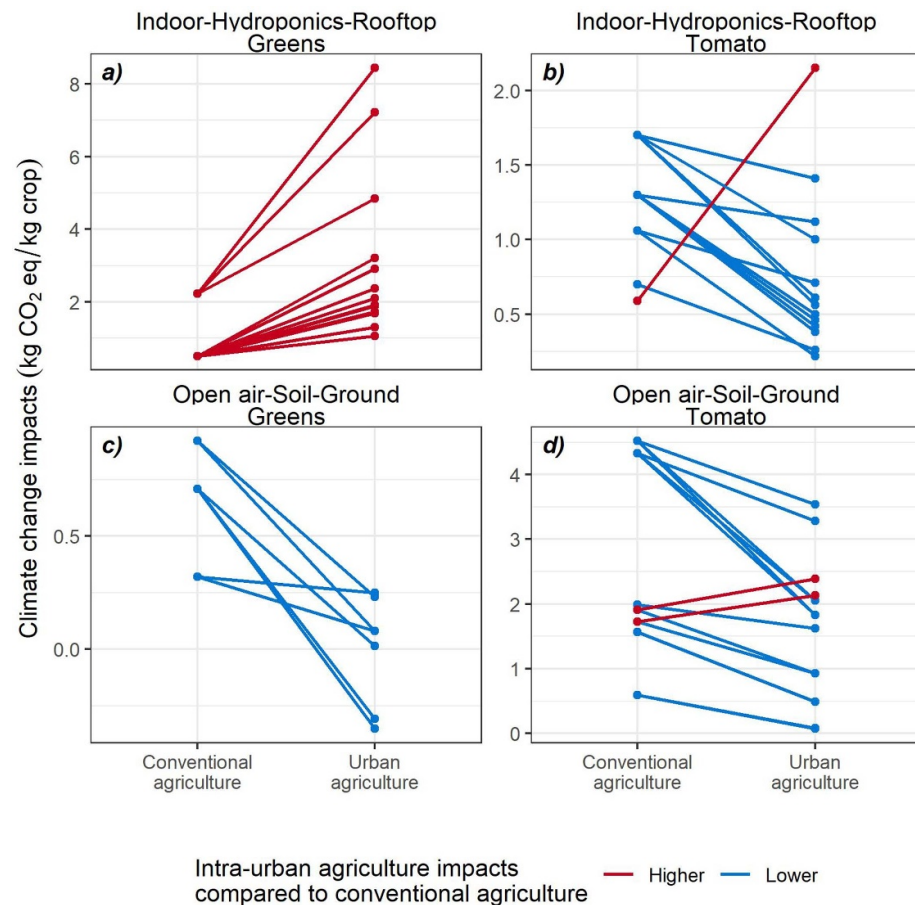
Figure 9. In many papers in this review, climate change impacts of the studied UA systems were compared to local, specific, conventional agriculture systems, for the same crop and physical farm/garden type. We summarized those comparisons here. Red lines represent comparisons where UA had larger impacts than the comparable conventional system, and blue lines represent the reverse. Note that the *y*-axis scales differ for each chart, representing the different local, specific conventional agriculture impacts used for comparisons in the literature.

of these comparisons were mixed, with UA sometimes performing better or worse than conventional agriculture across different crop types. Clune *et al* (2017) evaluated 122 LCAs with 633 climate change results for various fruits and vegetables, with a mass-based functional unit, and found that the majority of impacts were between 0.3 and 0.6 kg CO_2 eq. kg^{-1} crop. Among our 157 IUA results, most of the climate change impacts were between 0.3 and 4.0 kg CO_2 eq. kg^{-1} crop (lower and upper quantiles), which shows greater variability in results, and a tendency for larger impacts. PUA had lower climate change impacts than conventional agriculture for a mix of open-air crops (figure 10(a)). This may be due to the low-input nature of the PUA systems studied with rather simple LCAs here. For the IUA systems physically most similar to conventional agriculture—i.e. ground-based, open-air, soil-based systems—most of the climate change impacts were between 0.03 and 1.3 kg CO_2 eq. kg^{-1} crop. The mean of open-air systems here was more than twice the means of open-air fruits and vegetables from Clune *et al* (2017), but the medians were very close, again suggesting that in most cases the impacts were similar, but sometimes the

impacts were much greater in IUA systems. Lettuce only had similar impacts to conventional agriculture in open-air systems (using the median value), and had much larger impacts in all other comparisons (figure 10(c)). Tomatoes from IUA performed much better than leafy greens, and had lower impacts than indoor and overall conventional agriculture, using both the mean and the median (figure 10(d)). However open-air tomatoes in conventional agriculture had lower impacts than open-air UA tomatoes. In general, it appears that in most cases UA has similar impacts to conventional agriculture, but generated much larger impacts in a significant number of cases.

There were fewer examples of conventional agriculture climate change impacts by area than by mass of food, but we can nonetheless make some comparisons. Generally, we could expect conventional agriculture to have climate change impacts between 0.2 and 2 kg CO_2 eq. m^{-2} for cultivation of crops such as lettuce, tomato, onion, leek, pear, berries, cauliflower, and broccoli; for open-air, greenhouse, conventional, and organic systems (Meier *et al* 2015, Adewale *et al* 2016, Foteinis and Chatzisymeon 2016,
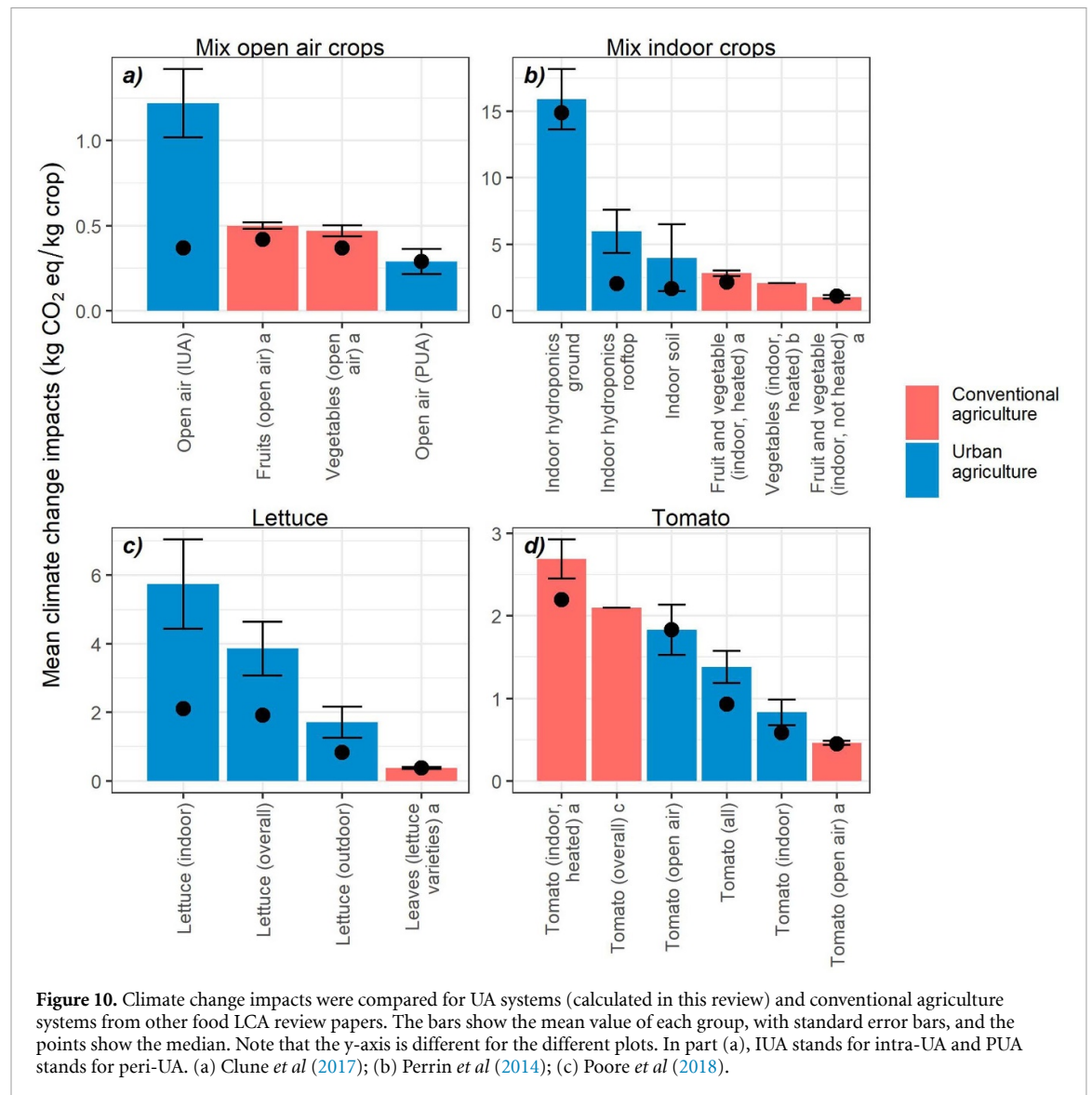
Figure 10. Climate change impacts were compared for UA systems (calculated in this review) and conventional agriculture systems from other food LCA review papers. The bars show the mean value of each group, with standard error bars, and the points show the median. Note that the y-axis is different for the different plots. In part (a), IUA stands for intra-UA and PUA stands for peri-UA. (a) Clune *et al* (2017); (b) Perrin *et al* (2014); (c) Poore *et al* (2018).
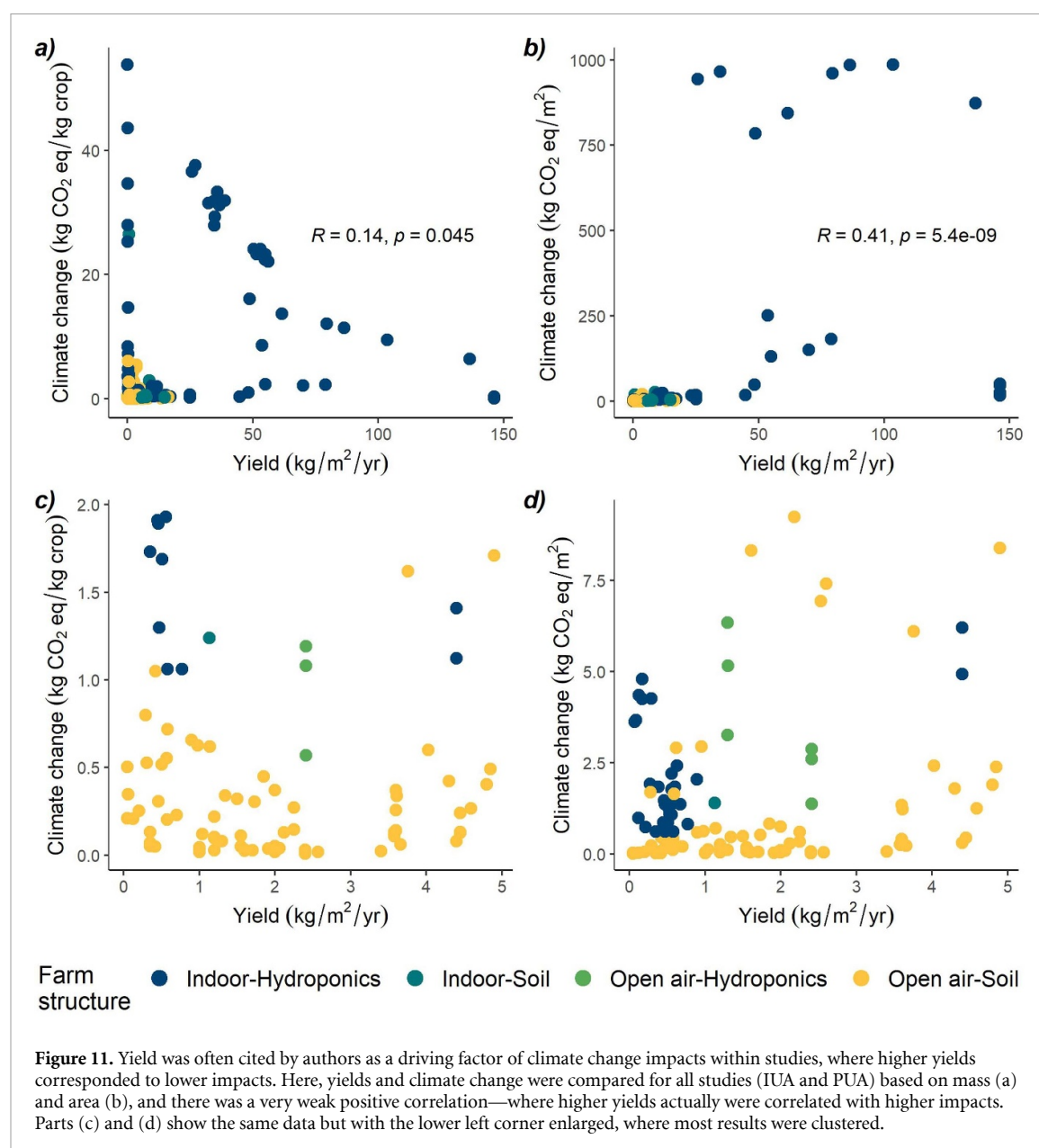
Poore and Nemecek 2018, Pereira *et al* 2021). The 112 IUA systems evaluated here had a global median of 4.7 kg CO_2 eq. m^{-2}, twice the upper bound of conventional agriculture. Open-air, soil-based, ground-based systems performed the best and were comparable to conventional agriculture, with a mean and median of 1.5 and 0.91 kg CO_2 eq. m^{-2}. The other physical system types all had larger mean and median impacts than the conventional agriculture range. The PUA systems had low impacts compared to conventional agriculture (0.51 ± 0.91 kg CO_2 eq. m^{-2}). Similar to the mass-based impacts, here UA has a mixed performance compared to conventional agriculture.

3.2.4.2. Features that largely affect climate change impacts

Our next objective was to explore what drove climate change impacts, and what made some UA systems more impactful than others. First, we evaluated a driving factor that was commonly identified by authors: crop yield (Kulak *et al* 2013, Sanyé-Mengual *et al* 2015a, 2015b, Goldstein *et al* 2016b, Cleveland

et al 2017, Dorr *et al* 2017, Martinez *et al* 2018, Pennisi *et al* 2019). Within these studies, comparisons between UA systems showed that those with higher yields had lower climate change impacts per kilogram of food, and impacts were overall very sensitive to changes in yield. Evaluating 199 paired yield and mass-based climate change impact values for IUA and PUA, we found a very weak (even negligible) correlation ($r = 0.14$, $p = 0.045$) in the opposite direction: higher yields corresponded to higher climate change impacts per kilogram (figures 11(a) and (c)). Although we could not conclude a strong positive correlation, we can rebuke the notion that there is an important negative correlation. Taking an area-based approach, there was a moderate positive correlation between crop yield and climate change impacts per m^2 ($r = 0.41$, $p = 5.4 \times 10^{-9}$) (figures 11(b) and (d)).

We can divide the yield and climate change impacts per kilogram of crop into quadrants, where the division between low and high yield is at 5 kg m^{-2} and for climate change is 2 kg CO_2 eq. kg^{-1} crop.

Figure 11. Yield was often cited by authors as a driving factor of climate change impacts within studies, where higher yields corresponded to lower impacts. Here, yields and climate change were compared for all studies (IUA and PUA) based on mass (a) and area (b), and there was a very weak positive correlation—where higher yields actually were correlated with higher impacts. Parts (c) and (d) show the same data but with the lower left corner enlarged, where most results were clustered.
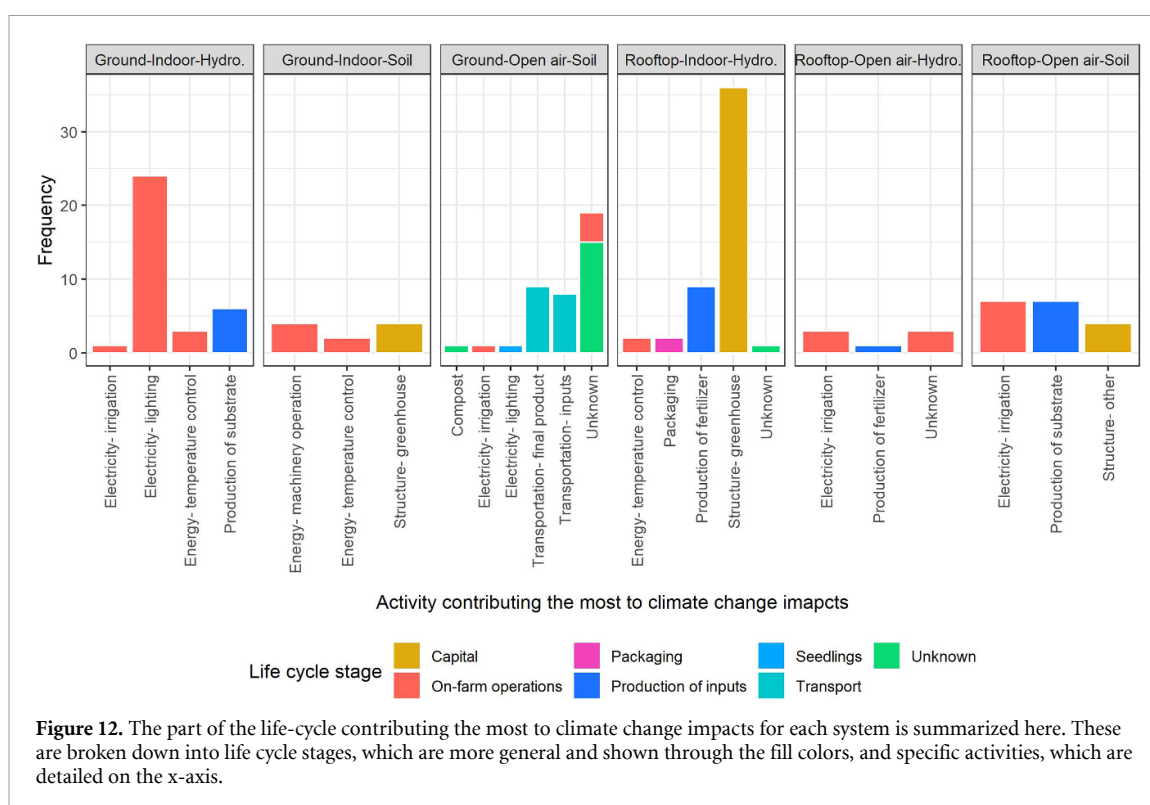
About half of the pairs fell in the low yield–low impact quadrant (47%). High yield–low impact systems made up 19% of the pairs, low yield–high impact systems 19%, and high yield–high impact systems 15%. High yield–low impact systems are particularly interesting because they would be the most viable way for UA to feed cities with reduced impacts. The 38 systems that were in this category came from 17 different papers, from all physical system types, several crop categories, all economic orientations, and both IUA and PUA. This suggests that there is no one UA type that optimizes food production and environmental impacts, but it is possible across a variety of types, management systems or other contextual factors which may be more important than system type (such as climate, objectives of the farm/garden, or constraints/opportunities based on the city's infrastructure).

Hotspot or contribution analysis was performed in most of the LCAs, and we evaluated these results to determine the most common, most impactful aspect for each system. We identified the single life-cycle stage (capital/infrastructure, seedlings, on-farm operations, production of inputs, packaging, and transport) and specific activity that accounted for the largest portion of climate change impacts in each system (figure 12). This was available for 90% of the systems studied, where it was identified by the authors, or could be interpreted using the data provided. We identified 15 different activities that were the most contributing to climate change impacts, which represents a substantial variation.

The most impactful stage was on-farm operations in 56% of systems where this information was available. This came mostly from energy consumption: for lighting, temperature control, and irrigation.

Figure 12. The part of the life-cycle contributing the most to climate change impacts for each system is summarized here. These are broken down into life cycle stages, which are more general and shown through the fill colors, and specific activities, which are detailed on the x-axis.

This was especially important for ground-based, indoor, hydroponics systems. Capital and infrastructure were the next most impactful (largest contributor of impacts in 20% of systems), and were mostly related to greenhouse structure for rooftop indoor systems. Production of inputs was the third most impactful life-cycle stage (19%) and came from production of substrate and fertilizer. Transport was found to be the most impactful in 7% of systems, and contributed especially large impacts to ground-based, open-air, soil-based systems, probably because they had overall low impacts with few structural and operational inputs. Transportation of inputs to the farms and gardens appeared as most impactful with similar frequency as transportation of the product to consumers, which was surprising given the lack of attention to the former and the potentially over-stated focus on the latter. Waste treatment did not emerge as most impactful in any system, and plant biomass composting contributed 1%–15% of climate change impacts (Sanjuan-Delmás *et al* 2018, Boneta *et al* 2019). Indoor soil-based and open-air hydroponics systems did not have as many results, but followed similar trends where greenhouse structure and on-farm energy use were highly impactful.

This breakdown of impact sources by life-cycle stage was similar to that of conventional agriculture, where most impacts usually come from the farm stage, direct energy use, and production and use of inputs (notably fertilizer for conventional agriculture) (Poore and Nemecek 2018). However, some activities emerged here as highly impactful which are not usually seen in conventional agriculture LCAs, including substrate production and transport of inputs. Conversely, direct N_2O emissions resulting from mineral fertilizer application is often a major source of climate change impacts in conventional agriculture, but did not appear important for UA. This was because mineral fertilizer was not often used on the farm, or was not included in the LCA. Inclusion of these direct emissions in the UA LCAs was often inconsistent and not transparent, but in some cases contributed 5%–12% to climate change impacts for the RTG hydroponics systems in Barcelona (Llorach-Massana *et al* 2017, Corcelli *et al* 2019).

For PUA (which is not shown in figure 12), on-farm operations were also the most impactful stage—specifically on-farm energy use for farm machinery and emissions from manure application. Production of inputs was the second most impactful category, from producing fertilizer and compost. According to the results of these contribution analyses, PUA appears much more related to conventional agriculture, where on-farm fuel use and fertilizer overwhelmingly emerge as the most impactful part of the life-cycle.

The physical setup of a farm appears to be a strong determining factor for the climate change impacts. Indoor farms require the burden of large material inputs for a greenhouse, container, or other artificial indoor environment, plus energy inputs for operation, and both often appear as the most impactful part of these systems. Despite the resulting increased

yields, this type of farm can still come with substantial impacts. This trend is often seen in conventional indoor agriculture as well (Perrin *et al* 2014, Clune *et al* 2017). However, the large variation in impacts of these systems, evidenced by the large standard deviations, suggest that there is real potential for improvement. In contrast, open-air, soil-based systems have the potential to be low-input, and even benefit from positive impacts such as uptake of large amounts of urban organic waste or sequester carbon in the soil. We hypothesize that the variation in impacts among different crops was more a reflection of the physical systems they were grown in than the crops themselves.

Finally, the carbon intensity of electricity grids can strongly influence the climate change impacts of energy-intensive indoor systems. Seven studies modeled different countries' electricity grids, or simulated energy provisioning from only renewable sources, and usually found profound differences in climate change impacts (up to a factor of 8), highlighting the inextricable nature of food and energy systems (Goldstein *et al* 2016b, Kikuchi *et al* 2018, Romeo *et al* 2018, Martin and Molin 2019, Martin *et al* 2019, Weidner and Yang 2020, Dorr *et al* 2021).

3.3. Quality and consistency of LCAs reviewed

A weakness of the body of literature was the lack of primary data from actual, functioning UA systems. This emerged when average values for conventional agriculture were taken from the literature and used for UA, which was a regular source of inventory data. For example, data from conventional agricultural inputs and yield were used for a recently established urban farm where production data were not yet available (Kulak *et al* 2013). Similarly, when authors focused on comparing one aspect of UA to conventional agriculture, they assumed agricultural inputs and yield were the same for both systems, and modeled only differences between certain aspects, such as greenhouse material and energy use (Torres Pineda *et al* 2020) or transport logistics (Oliveira *et al* 2021). Making such assumptions and using data from LCAs of similar systems is common practice in LCA, since the method is highly data-intensive, and using average values can make results more generalizable, but we argue that it is not appropriate here. Indeed, a main motivation for much of this research is to evaluate the specificities of UA (in contrast to conventional agriculture), so using inventory data from the same types of systems does not meaningfully discriminate between the two. Furthermore, many types of UA are immature, heterogenous, and regularly changing. Therefore, little is known about growing practices in UA, and it is premature to assume that they function the same as conventional farms. A review by Weidner *et al* (2019) that evaluated UA's potential to feed cities similarly found that most yield data did not come from actual UA case studies (75% of studies),

and in many cases came from conventional agriculture literature values (40%).

Additionally, many of the case studies in this review were in research settings, which allow for control of the physical setup of the farm, but eliminate the socio-economic aspects, real-world constraints, and human element of UA. Furthermore, they often focus on innovative systems or management techniques that may not be representative of typical UA systems, and which can be far from optimized due to their novelty. The conditions of UA may be more accurately represented if data are collected from functioning UA systems. This requires great effort from both researchers and practitioners, where the latter may lack motivation and time to commit to rigorous data collection. Therefore, on the other hand, developing research-oriented systems such as the iRTG in Barcelona can provide rich sources of high-quality data with the possibility to evaluate modifications in management.

Large standard deviations in many of these results challenge the consistency and quality of the literature. First, this high variability may result simply from inherent diversity in how UA systems are set up and operated. This review covered 259 diverse production systems from 88 farms and gardens, and few actual replicates (with the same physical form, purpose, and crop choices) were found. Second, this variability may arise from differences in LCA methods and choices, which is a near universal challenge for LCA, but perhaps especially relevant for this new application to a complex and diverse activity. Indeed, aspects were often explicitly excluded or not mentioned when they seemed relevant, such as:

- composting (as a material input and for treating farm biowaste),
- production and end-of-life of growing media/substrates,
- packaging,
- structural reinforcement of buildings with rooftop UA,
- transport to market or consumer,
- the nursery stage,
- food waste from UA,
- direct emissions from nutrient applications, and
- delivery of inputs to the farm.

Similarly, external consequences (see figure 1) in the form of avoided or 'positive' impacts were treated very differently with different effects on the results, and included:

- carbon sequestration in soils, substrate and compost,
- avoided agricultural land use (and possibly conversion to another land use),
- avoided production of mineral fertilizer when using or producing compost,

- avoided municipal organic waste treatment when using or producing compost or other organic waste/byproducts, and
- avoided municipal wastewater treatment by capturing run-off water.

Overall, this research topic is in its early stages. A relatively small number of LCAs were evaluated here, compared to LCAs of well-understood systems like energy, or even agricultural systems like wheat, where thousands of systems have been evaluated and results converge within a much narrower range (Poore and Nemecek 2018). LCAs of such agricultural systems have been done for nearly three times as long as UA LCAs (three decades), and great effort has been made to establish frameworks to ensure methodological consistency and generate meaningful results (Andersson *et al* 1994). Such work has not yet been done for UA, but would help to bring consistency to this topic.

LCA practitioners are increasingly calling for more holistic sustainability evaluations of life-cycle impacts, notably by including economic and social aspects. Although this review focused on environmental aspects, we must acknowledge that inclusion of other dimensions of sustainability will improve the relevance and decision-making potential of LCAs. This is especially true for UA, where socio-economic objectives are often emphasized. Among the papers in our review, six also performed life-cycle cost assessments (LCCAs), which evaluate the economic costs throughout the life-cycle of a product (Sanyé-Mengual *et al* 2015a, 2015b, Dorr *et al* 2017, Kim 2017, Pennisi *et al* 2019, Zhen *et al* 2020), and a review on LCCAs of UA described this in more detail (Peña and Rovira-Val 2020). Other non-life-cycle methods used to evaluate the economic dimension were cost-benefit analysis (Sanyé-Mengual *et al* 2015a, Pérez-Neira and Grollmus-Venegas 2018, Hu *et al* 2019, Liang *et al* 2019), economic efficiency (using profit as a functional unit) (Hu *et al* 2019, Rufí-Salís *et al* 2020a), and simple economic accounting indicators (Caputo *et al* 2020, Ledesma *et al* 2020). About two-thirds of the papers did not address economic aspects. Only two papers quantified social aspects: Fisher (2014) who evaluated labor hours and value, and Ledesma (2020) who scored indicators such as potential education benefits and safety risk during construction. The relative absence of the economic and social dimensions did not diminish the quality of environmental LCAs themselves, but diminishes the strength of this research topic as a whole (environmental assessment of UA) to evaluate the sustainability of UA.

3.4. Limitations
Consolidating and comparing LCAs is always a challenge because of differences in methodology, which may render results incomparable. Since LCA has only recently been applied to UA, we found large variability in the methodological choices, which lead to inconsistencies. For example, different functional units were used, and although we were able to convert them to a common 'kilogram of crop produced', the comparison of different food products is like comparing apples to oranges. Differences in system boundaries, (such as cradle-to-farm gate, to market, and to consumer) and in the inclusion or exclusion of uniquely UA processes (impacts of substrate production, certain avoided processes, building reinforcement) led to inconsistencies at the system modeling level.

A limitation to this meta-analysis was that papers included varying numbers of production systems, from 1 to 54 per article. Half of the papers evaluated only 1–3 production systems. Papers that evaluated many systems, as a result of variations in production or system modeling of one farm, had a large influence on the meta-analysis results. Examples include Caputo *et al* (2020), who evaluated 54 PUA open-air systems, Rufí-Salís *et al* (2020a) with 25 production systems from an indoor hydroponics RTG, and Pennisi *et al* (2019) with 20 indoor hydroponics systems. This may be especially important in this application where methodological choices between papers were rather inconsistent. Similarly, a large number of cases came from the same iRTG at the Universitat Autònoma de Barcelona in Spain (11 papers total, 8 papers in the meta-analysis, 44 systems, 17% of the systems evaluated), so the results were largely influenced by the material and operational design of this greenhouse (Sanyé-Mengual *et al* 2013, 2015a, 2018, Llorach-Massana *et al* 2017, Sanjuan-Delmás *et al* 2018, Toboso-Chavero *et al* 2018, Corcelli *et al* 2019, Rufí-Salís *et al* 2020a, 2020b, Arcas-Pilz *et al* 2021, Muñoz-Liesa *et al* 2021).

A limitation within the literature evaluated, and therefore of this review, was that the sample of farms and gardens are not necessarily representative of UA. Indeed, many authors did not explain why they chose to work with a given case study, or why experimental systems in research settings were designed the way they were. Furthermore, it is not clear what the scope of UA is in most cities and countries, so it is impossible to know if this pool of case studies is representative. Considering that farmers and gardeners must agree to invest great time and effort to provide data for LCAs, we can assume that there may be bias due to convenience sampling.

4. Discussion

4.1. Takeaways on the environmental performance of UA
The prevailing takeaway of this review was that existing LCAs are not sufficient to draw strong conclusions about the environmental performance of UA, especially in comparison to conventional agriculture.

Table 2. Some key trends and findings are summarized here.

Element	Key emerging trends
Indoor systems	Higher yield, higher climate change impact, higher energy use. Energy for lighting and temperature regulation, and greenhouse structure, were large sources of impact. Most results for herbs, tomatoes, vegetables, and leafy greens. Results varied based on ground–rooftop setting.
Open-air systems	Lower yield, lower climate change impact, lower energy use. Larger range of important sources of impact.
Intra-urban agriculture	Larger range of production system types. Smaller range of crop types. More results from UA case studies.
Peri-urban agriculture	Less varied production system types (mostly open-air, soil-based, ground-based). Larger range of crop types. More results from the literature and from conventional agriculture.
Research systems	Higher yield, higher climate change impacts. Almost the only system type with very large impacts. High quality and reliable data, but innovative, sub-optimized, and unrepresentative systems often studied.
Tomatoes	Most frequently studied crop, appeared in 36% of papers. Studied in all production system types except for ground-based, indoor, soil-based systems. Yield and impacts varied widely by farm type. Performed better against some types of conventional agriculture.
Lettuce	Second most frequently studied crop, appeared in 26% of papers. Studied in all production system types except for rooftop, open-air, hydroponics systems. Yield and impacts varied widely by farm type. Performed worse against conventional agriculture.
Water use	Life-cycle water use results were not as widely available as climate change impact results. Direct water use (mostly irrigation) was available for about 25% of systems. Water use was often higher for UA than conventional agriculture, although results varied widely.
Energy use	Life-cycle energy use results (cumulative energy demand, CED) were not as widely available as climate change impact results. CED results in about 25% of systems reviewed. CED had a strong positive correlation with climate change impacts. Open-air, soil-based systems had the lowest CED.

Researchers may never be able to draw broad conclusions about the sustainability of UA given the sheer diversity of UA. Inconsistency in the application of LCA methods compound this challenge. We found large variations in climate change impacts, energy demand, water use, and food production, differing by a factor of up to 5000. Across a diverse profile of system types, crops choices, and economic orientations, UA demonstrated the potential for both extremely small and surprisingly large impacts and yields. Generally, it appears that UA can substitute conventional agricultural without increasing food system impacts. The summary of results here will serve as a useful reference for positioning impacts and resource use efficiency from future LCAs of UA.

Looking across the studies, we still found some key trends that will help guide future decisions around UA. These trends are summarized in table 2. Indoor systems had larger yields, but also larger climate change impacts (based on area and mass) than open-air systems. Energy use (for lighting and temperature regulation) and greenhouse structure were most impactful for climate change in indoor systems, which certainly helped achieve higher yields, but apparently not high enough to compensate for their added impacts. The larger impacts in some cases may be explained by the experimental or innovative nature

of these indoor systems, where conditions were suboptimal and large opportunities for improvements were found. Leafy green crops, especially lettuce, and basil had the largest yields and climate change impacts, although this probably reflected the indoor-hydroponic systems where they were often cultivated. Open-air and non-commercial systems had lower climate change impacts and yields. Many different aspects emerged as having large climate change impacts in these systems, from transportation to production of substrate to irrigation. A lack of studies including water use efficiency and energy demand precluded identifying trends for these indicators. The variation in results for similar systems may also suggest that management practices influence environmental performance as much as or more than physical setup (e.g. indoor vs outdoor).

These results put into question the ideal that UA will substantially change urban food systems by displacing conventionally produced food, while simultaneously reducing climate change impacts. The systems with the lowest climate change impacts were those that are generally not poised to transform how cities procure food: non-commercial, lower yield (between UA options, but actually similar to conventional agriculture), land intensive, open-air soil-based farms and gardens. These often take

the form of school gardens, home gardens, and community farms, whose objective is usually not solely to compete with conventional agriculture in substantial, efficient provisioning of food. Furthermore, fruit and vegetables (the most common outputs of UA) are not large contributors to cities' climate change impacts (food consumption overall contributing about 10%–20% of climate change impacts, and fruits and vegetables accounting for only a portion of that) (Goldstein *et al* 2017a, González-García *et al* 2021). Even the most optimized scenarios would likely not see large climate change reductions from replacing conventional food with UA at the city scale (Goldstein *et al* 2017b).

We propose maintaining a nuanced and realistic perspective when evaluating UA, acknowledging that different types will come with different benefits and impacts, and that UA is not a singular activity with universal advantages. Specific UA projects should be promoted based on their specific, actual objectives and expected outcomes, which can have great merits even if they do not reduce the climate impacts of urban food systems. Cases with an actual focus on producing large amounts of food with reduced climate change impacts can draw from our results to design systems with minimized impacts by focusing on common hotspots.

4.2. Recommendations for future research

As in most applications of LCA, one of the greatest challenges here appears to be inconsistent methodological choices and reporting. This topic deserves greater attention for meaningful advancement and consolidation of UA LCAs, but some basic recommendations can be made here. Overall, we recommend aiming for the 'optimal' UA LCA example described in figure 1. Specifically, first, authors should describe case studies in greater detail, especially detailing what makes a system UA, because there is a great diversity within the category of UA. Generally, the physical farm setup was rather well characterized, but socio-economic aspects, which are a fundamental and diverse dimension of UA, were not usually detailed. For example, information about destination of the products for self-consumption, neighborhood, regional, or national scales; importance of food sales to the farm or garden; socio-economic and biophysical links to the city; attention and effort towards promoting biodiversity and ecosystem services; and purpose of the system and motivations of farmers and gardeners, would help communicate a more holistic view of the system. Second, we emphasize the importance of choosing system boundaries that include post-farm processes, because that is an essential tenet of the life-cycle perspective, and it is especially relevant for UA where proximity to the consumer is a unique characteristic. Third, we recommend that authors share line-by-line inventories and LCA results for each component of studied UA systems, in the text

or in supplemental materials. Essential line-by-line information includes yield, direct water use, direct energy use, amount and type of inputs such as compost and fertilizer, distance and mode of transport for delivery of products, avoided processes or impacts, and seasonality. This is good practice for LCA in general, but it is especially important for such diverse systems as UA where little is known and the relevant components may vary. Finally, we recommend using multiple functional units in order to capture multiple dimensions of systems. Our results, and a large body of research comparing organic and conventional agriculture, shows that performance of agriculture varies when using mass-based and area-based functional units (Meier *et al* 2015, Van Der Werf *et al* 2020). Other functional units that may be relevant are nutritional indexes, economic output, ecosystem services, or quantified social outputs.

Furthermore, pursuing this area of research and performing more high-quality LCAs of UA is essential, because a relatively small number of cases were reviewed here. Simply collecting the data necessary for such LCAs, from actual UA case studies, is a valuable contribution to our understanding of how UA operates and what the outcomes are. LCAs of scaled-up UA at the city level, or at personal consumption scale, are important for putting these impacts in perspective, but they should be based on strong farm-level data of actual UA cases, which is currently lacking.

Finally, we encourage reflection on the purpose and direction of LCA of UA. LCA is oriented towards evaluating environmental impacts based on the efficiency of systems producing goods and/or services. Where UA is positioned to optimize growing food—in a focused, commercial, and more or less efficient manner—environmental LCA is an appropriate tool. However, this is often not the case. In many contexts for UA, food production is a shared or minor objective after more social objectives (Guitart *et al* 2012, Buehler and Junge 2016, Pourias *et al* 2016, Orsini *et al* 2020, Appolloni *et al* 2021). Here, it is not very relevant to evaluate the efficient use of inputs for growing food in UA and position it next to conventional agriculture or other urban land uses, or assess its capacity to substantially reduce impacts of an urban food system. At the same time, LCA only captures a fraction of what UA is. The full benefits of UA, including social objectives, and even numerous environmental dimensions, are fundamentally outside the scope of LCA.

5. Conclusion

Applying LCA to UA is still in its infancy, and thus far has evidenced a very wide range of outcomes for yields, water use, energy demand, and climate change impacts, across different physical setup, crops, and socio-economic orientations. This

evaluation framework clearly needs to be further strengthened and consolidated before it can guide the design and management of UA systems, and provide robust estimates for their performance. We identified initial trends and summarized baseline values across different UA profiles, but could not arrive at strong conclusions due to quality and consistency issues with the literature. As more and more references will become available, the methodological guidelines laid out in this review should help clarify trends and answer key questions, in particular regarding comparisons between different types of UA or the comparison to conventional agriculture. The outcomes of this review can shift the direction of and help improve LCAs of UA, and provide nuance to broader evaluations of the potential outcomes of UA.

Data availability statement

The data that support the findings of this study are openly available.

All data that support the findings of this study are included within the article (and any supplementary files).

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