Environmental performance of urban agriculture: how to apply life cycle assessment, and the knowledge and questions generated

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La performance environnementale de l'agriculture urbaine : comment appliquer l'analyse du cycle de vie, et les connaissances et questions générées

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ÉCOLE DOCTORALE
Agriculture, alimentation, biologie, environnement, santé (ABIES)

Titre : La performance environnementale de l'agriculture urbaine : comment appliquer l'analyse du cycle de vie, et les connaissances et questions générées

Mots clés : analyse du cycle de vie, agriculture urbaine, impacts environnementaux, systèmes alimentaires durables

Résumé : Le système alimentaire mondial a un impact considérable sur l'environnement et doit relever le défi de nourrir une population encore plus nombreuse et plus urbanisée dans les décennies à venir. L'agriculture urbaine (AU) est un type d'agriculture alternatif, qui peut présenter des avantages environnementaux et sociaux, et se manifeste sous une grande diversité de formes. Ces avantages et impacts environnementaux peuvent être modélisés par l'analyse du cycle de vie (ACV). L'application de l'ACV à l'agriculture urbaine est relativement récente et n'a pas fait l'objet des mêmes réflexions et adaptations méthodologiques que l'ACV d'autres secteurs. Dans ce projet de thèse, j'ai cherché à savoir 1) ce que l'ACV nous apprend sur la performance environnementale de l'agriculture urbaine, et 2) comment appliquer au mieux l'ACV à l'agriculture urbaine. J'ai effectué un examen et une méta-analyse des ACV de l'agriculture urbaine, et j'ai passé en revue la littérature sur le développement de l'ACV pour l'agriculture en général. J'ai réalisé l'ACV de neuf fermes et jardins urbains à Paris, en France, et dans la Bay Area, en Californie, aux États-Unis, et (avec le projet FEW-meter) j'ai mesuré les intrants et les extrants de 72 études de cas d'agriculture urbaine. J'ai résumé et généré des connaissances sur la performance environnementale de l'agriculture urbaine, et créé un cadre méthodologique pour améliorer la cohérence et l'exhaustivité des ACV de l'agriculture urbaine.

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Keywords: life cycle assessment, urban agriculture, environmental impacts, sustainable food systems

Abstract: The global food system causes massive environmental impacts, and faces the challenge of feeding an even larger, more urbanized population in the coming decades. Urban agriculture (UA) is a type of alternative agriculture, which may have environmental and social benefits, and comes in a large diversity of forms. These environmental benefits and impacts can be modeled with life cycle assessment (LCA). Application of LCA to UA is relatively recent, and has not undergone the same methodological reflections and adaptations that LCA of other sectors has. In this thesis project, I investigated 1) what LCA tells us about the environmental performance of UA, and 2) how best to apply LCA to UA. I performed a review and meta-analysis of UA LCAs, and reviewed literature on the development of LCA for agriculture in general. I did LCAs of nine urban farms and gardens in Paris, France and the Bay Area, California, USA, and (with the FEW-meter project) analyzed resource use and food production at 72 UA case studies. I summarized and generated knowledge on the environmental performance of UA, and created a methodological framework to improve consistency and completeness in UA LCAs.

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Résumé long

Le système alimentaire mondial a un impact considérable sur l'environnement et ne parvient pas à nourrir correctement une grande partie de la population. Les systèmes alimentaires et agricoles doivent relever le défi de nourrir une population encore plus nombreuse et plus urbaine au cours des prochaines décennies, tout en respectant les limites environnementales planétaires, qui sont déjà mises à rude épreuve. L'agriculture urbaine (AU) est une alternative, elle-même diverse au système conventionnel, qui peut présenter des avantages à la fois environnementaux et sociaux.

Les avantages et impacts environnementaux peuvent être modélisés par l'analyse du cycle de vie (ACV), une méthode qui modélise les multiples impacts environnementaux d'un produit ou d'un service tout au long de son cycle de vie. L'ACV devrait permettre de déterminer si l'agriculture urbaine peut atténuer les dommages environnementaux du système alimentaire, dans quelle mesure, et dans quelles conditions. L'ACV a été conçue et d'abord utilisée pour les systèmes industriels, qui sont très différents des systèmes agricoles. Une réflexion, un développement et une harmonisation considérables ont été nécessaires au cours des 20 dernières années pour améliorer l'ACV en fonction des aspects spécifiques des systèmes agricoles. L'application de l'ACV à l'agriculture urbaine est relativement récente et n'a pas fait l'objet des mêmes réflexions et adaptations méthodologiques que l'ACV des autres secteurs.

Dans ce projet de thèse, j'ai étudié à la fois les méthodes et les résultats des ACV de l'agriculture urbaine. Plus précisément, mes questions de recherche étaient les suivantes 1) En quoi l'agriculture urbaine se distingue-t-elle des autres systèmes de production et quelles sont les conséquences sur son évaluation par l'ACV ? 2) Que nous apprend l'ACV sur la performance environnementale de l'agriculture urbaine ? Et 3) Comment devrions-nous appliquer l'ACV à l'agriculture urbaine pour faire des évaluations plus cohérentes, plus complètes et plus adaptées ?

La première question de recherche a été principalement abordée par le biais de 72 études de cas de fermes et de jardins urbains dans le cadre du projet FEW-meter (Projet SUGI impliquant cinq pays dont la France et les Etats Unis.) Nous avons utilisé des méthodes participatives pour obtenir des données primaires sur la production alimentaire et l'utilisation des ressources pour divers types d'agriculture urbaine dans ces cinq pays. Cette question a également été abordée lors de mon travail personnel avec neuf fermes et jardins urbains à Paris, en France, et dans la région de San Francisco, en Californie, aux États-Unis, afin de recueillir des données primaires et de réaliser des ACV. Sur la base d'un échantillon total de 81 études de cas d'agriculture urbaine, j'ai ainsi été en mesure de faire quelques généralisations sur les modes de fonctionnement spécifiques de l'agriculture urbaine et de réfléchir à leurs implications pour les ACV. Parmi ces points, citons la variabilité particulièrement élevée des opérations en agriculture urbaine (liée à la variabilité de ses systèmes techniques), l'utilisation importante de compost et d'eau potable des villes (ce qui n'est généralement pas le cas en agriculture rurale), l'absence, dans le cadre des type d’agriculture urbaine investigués, d'engrais et de pesticides synthétiques et la consommation d'énergie à la ferme (ce qui est généralement important pour les ACV de l'agriculture rurale), la diversité extrêmement élevée des cultures et certaines relations inexpliquées entre des caractéristiques agronomiques de base, comme les apports en compost et en eau et le rendement.

Pour la deuxième question de recherche, j'ai effectué un examen et une méta-analyse des ACV de l'agriculture urbaine afin de résumer les impacts et l'utilisation des ressources pour de multiples types d'agriculture urbaine et déterminer les tendances entre ces types. Les ACV que j'ai effectuées sur neuf fermes et jardins urbains à Paris, en France, et dans la Bay Area région de San Francisco, en Californie, aux États-Unis, ont également largement contribué à mes réponses à cette question. En général, les impacts de l'agriculture urbaine sur le changement climatique étaient similaires à ceux de l'agriculture conventionnelle rurale, mais ils étaient beaucoup plus variables. La consommation d'eau était beaucoup plus importante que prévu dans certains cas d'agriculture urbaine, mais là encore, les variations étaient importantes. Entre les types d'agriculture urbaine, les systèmes plus professionnels, commerciaux et de moyenne à haute technologie avaient tendance à avoir des impacts plus faibles par kilogramme, mais plus élevés par m2, que les systèmes sociaux à faible technologie. Cependant, certains types de systèmes de haute technologie, avec des intrants importants mais aussi des rendements élevés, avaient encore des impacts importants par kilogramme. La variabilité des impacts entre et au sein des types d'agriculture urbaine a rendu difficile l'établissement de tendances. Les sources importantes d'impact lié au changement climatique varient selon le type de système : pour les systèmes en intérieur, l'utilisation de l'énergie est souvent la plus importante, pour les serres sur toit, c'est la structure de la serre, et pour les autres systèmes, les sources d'impact sont variées. La fourniture d'intrants à la ferme s'est avérée impactante, au même titre que les impacts de la livraison du produit au marché/consommateur, ce qui suggère un compromis à trouver dans la localisation géographique de l'agriculture urbaine. Dans mes neuf études de cas d'ACV de l'agriculture urbaine, les processus qui ont largement contribué à plusieurs catégories d'impact sont l'infrastructure, l'irrigation, le compost et la tourbe pour les semis. J'ai identifié les paramètres sensibles et les décisions de modélisation du système, comme la répartition des impacts entre le produit de compostage et le service de traitement des déchets, la durée de vie de l'infrastructure et du substrat, l'inclusion du transport des clients vers la ferme pour les ventes directes ou les systèmes de cueillette, l'inventaire du compost utilisé et l'inclusion de la séquestration du carbone à partir du compost.

L'ensemble de ces recherches a permis de répondre à la troisième question, à savoir le développement méthodologique et la création d'un cadre pour les ACV en agriculture urbaine. Tout d'abord, l'analyse documentaire des ACV de l'agriculture urbaine et rurale a montré que la recherche sur l'ACV de l'agriculture urbaine est relativement naissante et immature. Une réflexion méthodologique était nécessaire pour apporter de la cohérence et de l'exhaustivité, et pour (re)cadrer l'intérêt/le rôle de faire des ACV en agriculture urbaine. À partir d'une analyse documentaire, de mesures empiriques des intrants et des extrants de 72 études de cas d'agriculture urbaine et de mes propres ACV effectuées sur neuf études de cas, j'ai élaboré un cadre méthodologique visant à améliorer les ACV d'agriculture urbaine. L'objectif de ce cadre est d'améliorer la cohérence et l'exhaustivité des études et de l'ensemble de la documentation. Le cadre comprend des suggestions de questions/objectifs auxquels les ACV de l'agriculture urbaine peuvent chercher à répondre, dont certaines ont été trouvées dans la littérature et d'autres sont nouvelles. J'ai ensuite abordé dix sujets qui sont propres à l'agriculture urbaine et qui posent des problèmes pour les ACV. Pour chaque sujet, j'ai décrit le défi, présenté des exemples de la façon dont il a été abordé dans les ACV de l'agriculture urbaine ou de systèmes similaires pertinents, puis recommandé la façon de l'aborder à l'avenir. Les sujets abordés sont la diversité des cultures, la multifonctionnalité, la disponibilité des données, la modélisation des systèmes de compostage hors ferme, la modélisation des systèmes de compostage à la ferme, la séquestration du carbone, les facteurs d'émission de GES du compost, la création du substrat, le transport et la livraison, et la variabilité des formes de l'agriculture urbaine. Le cadre se termine par une section présentant les orientations de la recherche sur les ACV en AU, qui visent à soutenir à la fois la recherche et l'application.

J'espère que les résultats de ce projet de thèse aideront le domaine des ACV de l'agriculture urbaine à passer d'un sujet jeune, incohérent et peu clair à un sujet de recherche plus mature, riche et harmonisé. Nous pouvons mettre en perspective cette thèse en considérant l'objectif initial de ce travail, qui a été rapidement redéfini : créer un outil d'ACV simplifié pour l'agriculture urbaine. En réalisant que les ACV de l'agriculture urbaine n'en étaient qu'à leurs balbutiements, nous avons réalisé qu'il n'était pas opportun de développer un tel outil avant d’avoir fait ce travail préalable de cadrage et d’identification des problématiques liées à cette thématique. Il n'y avait tout simplement pas assez de connaissances disponibles pour faire les simplifications nécessaires à la création de l’outil. Au lieu de cela, j'ai creusé le sujet et révélé ses complexités, ses problèmes et ses lacunes, afin d'aider à trouver une meilleure voie à suivre. Espérons que grâce aux connaissances générées par le travail empirique, aux informations résumées dans la littérature et au cadre proposé pour harmoniser les travaux futurs, cette thèse améliorera notre compréhension de la performance environnementale de l'agriculture urbaine.

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CED	Cumulative energy demand
ES	Ecosystem service
FEW	Food-energy-water
FU	Functional unit
GHG	Greenhouse gas emissions
iRTG	Integrated rooftop greenhouse
IPCC	International Panel on Climate Change
ILCD	International Reference Life Cycle Data System
ISO	International Standardization Organization
IUA	Intra-urban agriculture
LCA	Life cycle assessment
LCCA	Life cycle cost assessment
LCI	Life cycle inventory
LCSA	Life cycle sustainability assessment
NR-CED	Non-renewable cumulative energy demand
PUA	Peri-urban agriculture
PCA	Principal component analysis
PEF	Product environmental footprint
RTG	Rooftop greenhouse
S-LCA	Social life cycle assessment
SETAC	Society of Environmental Toxicology and Chemistry
SCG	Spent coffee grounds
SMS	Spent mushroom substrate
UA	Urban agriculture
UNEP	United Nations Environmental Program
USDA	United States Department of Agriculture
UGI	Urban green infrastructure

Introduction

I. Introduction

I.1. Food system environmental issues

Humanity faces major environmental problems. After decades of emitting greenhouse gasses in the industrial revolution, we are changing the global climate, threatening a cascade of ecological and societal transformations (IPCC, 2022). The sixth mass extinction is underway, caused this time by humans, threatening biodiversity and the precious ecological systems that depend on it (IPBES, 2019). We face the prospect of running out of the limited resources that society depends on, such as fuel and minerals, while also disturbing biogeochemical cycles (FAO, 2008; IEA, 2021).

Agriculture is a major contributor to these crises. Enteric fermentation in cows causes a continuous leak of potent greenhouse gasses. Expansion of agriculture into 'natural' lands, notably in tropical forests, destroys the carbon sinks that these ecosystems have provided for millennia. Massive amounts of energy (usually from fossil fuels) are used to transform atmospheric nitrogen into a mineral form to use as fertilizers and grow crops larger and faster. Overall, the food system is estimated to account for a quarter to a third of global greenhouse gas emissions (Crippa et al., 2021; Poore and Nemecek, 2018). Much of the nitrogen from fertilizer remains in the environment rather than consumed in food, leading to pollution of soil, water and air (Galloway and Cowling, 2002). Limited stocks of phosphorus are extracted from the earth and applied to soils around the world to promote crop growth. Toxic pesticides are applied to crops and indiscriminately poison non-targeted insects and plants. The presence of agriculture disrupts wild animals' habitats and leads to population decline or extinction. All of this to produce food, of which about one third is not consumed (FAO, 2019; UNEP, 2021).

As hubs of people and therefore food consumption, cities drive the need for agriculture (Grimm et al., 2008). Food is one of the main resource flows into cities, and accounts for a large part of the environmental impacts of consumption in cities (Beylot et al., 2019; Ivanova et al., 2016). In 2018, 55% of the world's population live in cities, and this is expected to reach about 70% by 2050 (United Nations, 2018). Millions in cities in developing and developed economies face food insecurity, while large amounts of food are wasted. At the same time, the global population is expected to increase from 7.7 to 9.7 billion by 2050, raising the question of how to provide more food without exacerbating the already dire environmental pressures from agriculture (United Nations, 2019).

I.2. Urban agriculture as an alternative

Across several dimensions, current food systems are unsustainable. Urban agriculture (UA) is an alternative that may offer some remedies to the ills of the food system. UA is both an alternative form of food production, and alternative urbanism, and the solutions it proposes are as numerous as the visions of it. Simply, UA is defined as producing food in cities. A more comprehensive definition is provided by Mougeot et al. (2000):

> "Urban Agriculture is an industry located within (intra-urban) or in the fringe (peri-urban) of a town, a city or a metropolis, which grows and raises, processes and distributes a diversity of food and non-food products, (re)-using largely human and material resources, products and services found in and around that urban area and in

> turn supplying human and material resources, products and services largely to that urban area."

There is a long tradition of food production in and around cities (Barthel and Isendahl, 2013; Howe and Wheeler, 1999). Agriculture was included in the city plan for Kyoto, Japan in the eighth century (Yokohari et al., 2000), and Mayan and Aztec cities featured smallholder, intensive agriculture within the city limits (Isendahl and Smith, 2013). Up until 60 years ago, and for over a millennium, the area surrounding Paris was largely responsible for feeding the city (Billen et al., 2009), and vegetables were cultivated within Paris (on 1/6th of the city's area) from the twelfth to nineteenth centuries (Lawson, 2016). As industrialized cities became denser and expanded their limits, agriculture was pushed out, alienating urban residents from nature and food production (Howe and Wheeler, 1999). This rupture is referred to as the metabolic rift, where consumers become disconnected from producers, and flows between the two become imbalanced (Dehaene et al., 2016; McClintock, 2010). Allotment gardens in cities in the UK and Germany have been prevalent since the nineteenth century (Smit et al., 2001). The garden city, proposed by the influential urban planner Ebenezer Howard, envisioned a utopian proximity and harmony between cities and agriculture in 1898, as a response to his experiences of overcrowded, unsanitary Victorian London (Howard, 1898). More recently, during both World Wars, urban agriculture returned to cities in England to provide produce to cities under blockades in *War Gardens.* In the US this took the form of *Victory Gardens*, aimed at alleviating food insecurity after diverting food to Europe (Mok et al., 2013). Cuba got about 60% of its vegetables from UA, in response to being cut off from global markets during the collapse of the soviet bloc between 1989 and 1993 (Altieri et al., 1999; Lawson, 2016).

UA is experiencing a renewed interest in the Global North (Reynolds and Darly, 2018). Although there is no global census on UA, it is clearly developing more in cities in the Global North, with promotion by local governments and national plans, and increasing interest from the research community (Smit et al., 2001). Part of the renewed interest is the awakening of urbanites to the role of the food system in environmental and health crises. In California, alternative food movements have shifted from a focus on rural, 'producer' issues in the 1970s and 80s—such as social justice for rural migrant farmers, and the organic movement—to a focus on the urban 'consumer' issues (Allen et al., 2003). Mainstream books and movies, such as *Omnivore's Dilemma* and *Food, Inc.*, (Kenner, 2008; Pollan, 2007), have introduced massive audiences to critiques of the global industrial food system, and brought the alternative food movement to the masses (Mok et al., 2013). In the USA, the First Lady Michelle Obama popularized food issues by planting a backyard garden at the White House in 2009, and campaigns related to healthy eating and gardening ("Let's Move," 2022; Mok et al., 2013). The current iteration of UA sees longstanding forms of UA—such as allotment gardens, school gardens, home gardens, and community farms—joined by urban farms, rooftop gardens, rooftop greenhouses, and indoor hydroponics vertical farms (Mok et al., 2013; Reynolds and Darly, 2018). Another feature of the modern iteration of UA is its position in the city center, as opposed to next to or near cities (Reynolds and Darly, 2018).

Types of UA are defined by both their technical form and motivations, and include community gardens, urban farms, vertical farms, home gardens, allotment gardens, school gardens, among others. Garden usually denotes a non-commercial system, while farms may be more commercial and/or professional, although it is a spectrum with many systems situated in the middle (Reynolds and Darly, 2018). The context and meaning of UA varies across cities, based on the culture and history (Smit et al., 2001). Three major axes of UA are its agricultural, urban land use, and social function (described below).

I.2.1. Urban agriculture as an agricultural activity: alternative food system

Food production is necessarily an aspect of all UA projects. Some claim that food production is a relatively unimportant outcome of UA in the Global North (due to physical limits, or objectives that contradict productivity), while others maintain that UA is a highly productive activity, among its other functions (Orsini et al., 2020; Pourias et al., 2015a; Siegner et al., 2018). The level of production of UA has been shown to vary, with high or low levels of self-sufficiency at the individual and the city level (Weidner et al., 2019). This variability is due to actual differences in UA projects, and due to uncertainty in the productivity/efficiency of UA. The main function here is not only contributing to the local food supply, but specifically providing fresh and healthy food to those who may otherwise not have access to it (Artmann and Sartison, 2018). UA is an alternative type of agriculture that adapts to constraints and limitations in the city. It uses traditional agricultural techniques and develops new ones, such as growing food on organic waste products instead of in the soil (Gomez Villarino et al., 2021; Grard et al., 2018). It usually involves organic practices, and is seen as an environmentally virtuous form of agriculture, although this is not always the reality (Guitart et al., 2012; Santo et al., 2016). Urban agroecology is proposed as UA with an emphasis on agroecology, which includes elements such as diversity, synergies, efficiency, resilience, recycling, co-creation and sharing of knowledge, social values, and responsible governance (FAO, 2018; Gomez Villarino et al., 2021). UA faces unique issues around access to land (greenfields or otherwise), soil quality, shading from buildings, and long-term viability (due to economic and land tenure issues) (Santo et al., 2016).

I.2.2. Urban agriculture as a tool for urban sustainability: green urban land use

Artificial spaces are increasingly dominating, with manufactured materials such as steel and concrete now making up more mass on the planet than living things (Elhacham et al., 2020). Cities are especially dominated by their built environment. The vision of cities as a type of ecosystem—the urban ecosystem—is relatively recent (Grimm et al., 2008). Cities are no longer recognized as separate non-living spaces, but as a part of a landscape. Reintroducing ‘nature’ into cities through urban green infrastructure (UGI), such as UA, is an increasing priority with benefits to well-being of urban residents and to local ecosystems (detailed below). UA is one choice among many to allocate scarce urban space for UGI, among other nature-based solutions such as parks, urban forests, or water bodies. Other alternative non-green urban land uses compete with UA, such as renewable energy production and innovative cooling materials (Croce and Vettorato, 2021).

UGI such as UA can improve cities’ climate resilience and sustainability through lowering air temperatures, managing stormwater runoff, and insulating buildings (Ferreira et al., 2018; Russo et al., 2017). It can support urban biodiversity, which surprisingly plays an essential role in conservation of threatened species (Aronson et al., 2017; Kühn and Klotz, 2004). Twenty two percent of occurrences of endangered plants in the US are found in metropolitan areas, and 30% of all threatened species in Australia can be found in their cities (Ives et al., 2016; Schwartz et al., 2002). Within the last year, headlines were made when an orchid, previously thought to be locally extinct, was found on the green roof of a bank in downtown

London, England; and when a rare bird not seen locally in nearly ten years was found on the rooftop park of the Salesforce building in San Francisco (BBC News, 2021; Robertson, 2021). UGI also contributes to urban renewal, mental and physical health, social cohesion, and wellbeing (Coutts and Hahn, 2015). While UA is one among many UGI solutions, it may represent a particularly valuable one thanks to its multifunctionality (Artmann and Sartison, 2018; Gomez Villarino et al., 2021).

I.2.3. Urban agriculture as a social activity: tool for well-being and social change

The benefits that UA directly brings to people in terms of wellbeing, recreation, community building, civic engagement, political/radical activism, and public health are for many the main point of UA in the Global North (Feola et al., 2020; Gomez Villarino et al., 2021; Santo et al., 2016; Siegner et al., 2020). Although some dismiss UA as a feel-good, greenwashed, bourgeoise activity, its contributions to food security and community development may be serious, for marginalized and non-marginalized communities alike (Mok et al., 2013). UA may aim to address the racial and economic injustices in the food system, where disenfranchised urbanites face higher rates of food insecurity and lower availability of healthy foods (Franco et al., 2008). It may offer job training for marginalized people, such as those who were formerly incarcerated, and improve wellbeing at the neighborhood scale by uplifting otherwise neglected communities (Poulsen, 2017). UA may foster social interactions and community building in vulnerable population groups such as the elderly, those recovering from natural disasters, unemployed, low-income, and recovering patients in hospitals (Artmann and Sartison, 2018). In these cases of UA, food is the vehicle that urban farms/gardens use to improve communities (Poulsen, 2017).

I.2.4. An integrated solution

In most cases, UA is simultaneously an alternative agricultural, urban, and social system. This multifunctionality is one of its greatest strengths—the ability to supply multiple benefits towards sustainable cities and sustainable food systems (Gomez Villarino et al., 2021). UA is a component of several different visions of alternative systems— alternative urbanism and alternative food systems—and can adapt to local needs. It may be part of the solution to several problems simultaneously (Dehaene et al., 2016). Indeed, it may be difficult to justify using scarce, expensive urban land based on the food production function alone (Lovell, 2010). UA may demonstrate the strong multifunctionality that agriculture is supposed to inherently display, but has lost through the industrialization and commodification of agriculture (Lovell, 2010).

I.3. Environmental assessment of UA

While remaining conscious of the many dimensions of UA, this thesis focuses on its environmental dimension. Food production, resource use, and environmental benefits and impacts occur in UA even when environmental aspects are not a main priority or motivation. The discourse around UA overwhelmingly promotes environmental benefits, including less transport, packaging, and processing of produce; reusing urban waste, such as organic waste for compost and greywater for irrigation; and benign production practices such as organic or agroecological production (Lovell, 2010). More recently, uncertainty and critiques have emerged, suggesting that UA may be relatively inefficient and resource consuming due to small economies of scale (McDougall et al., 2019; Mok et al., 2013), and that the importance of proximity to the consumer and reduced ‘food miles’ may be overexaggerated (Born and Purcell, 2006; Edwards-Jones et al., 2008; Goldstein et al., 2016a). Environmental assessments are necessary to test assertions about the environmental performance of UA.

I.3.1. Overview of evaluation methods

Many environmental assessment methods have been used to draw valuable insight about UA (Feola et al., 2020). Some of these methods, and examples of their application to UA, are summarized in Table I.3.1.

I.3.2. Life cycle assessment

While many methods described in Table I.1 provide valuable insight into the operations and environmental benefits and impacts of UA, they fail to provide a systemic view. Systems thinking is necessary to develop sustainable food systems because it allows for consideration of tradeoffs, feedback, and interdependence between components (FAO, 2015; Williams et al., 2017). Systems thinking involves understanding relationships between multiple components, between components and the whole, and between the whole and the system in which it is embedded (which is itself just another whole) (Amissah et al., 2020). For example, in agriculture it is insufficient to consider improving water use efficiency looking only at water use and food produced, because other critical mechanisms are at play such as climate, crop physiology, nutrient supply, and pests (Bindraban et al., 2010). Similarly, in the food system, it is insufficient to consider only agricultural production and not the distribution system that follows (Andersson et al., 1994). Systems thinking provides a more holistic view of processes, and can be scaled to various levels, from the molecule to the sector to the world. Life cycle thinking represents a type of system thinking, as it considers all aspects of a process or products from its raw material extraction through to its end-of-life waste treatment (Figure I.3.2). Life cycle thinking helps us to recognize that processes and decisions are not isolated, and instead influence a larger system. This helps to improve entire systems in their multiple parts and complexity, rather than addressing single components which may shift burdens/impacts within a system (UNEP, 2004, p. 2004). The central method in the life cycle approach is life cycle assessment (LCA).

LCA is defined as the “compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle”—that is, from its creation through use and end of life (ISO, 2006a). The first LCA was done by the Coca-Cola company in 1969, with a study comparing resource use and waste management of different packaging options (Huppes and Curran, 2012). These early years of LCA, from the 1970s to 1990s, saw little scientific interest in the methodology, and were characterized by a large variety of practices, lack of common theoretical framework, and divergent results, leading to an unreliable analytical tool (Guinée et al., 2011; Heijungs and Guineé, 2012). The ‘decade of standardization’, from 1990-2000, saw a convergence in LCA practices thanks to coordination efforts by the Society of Environmental Toxicology and Chemistry (SETAC) to unite scientists; LCA users and practitioners to harmonize practices; and the standardization work of the International Organization for Standardization (ISO), culminating in the first ISO report

Method	Description	Examples for UA
Agronomic indicators	Empirical measures of functioning and performance that simplify a system.	Measurements at the farm-scale of water use (Csortan et al., 2020; Dalla Marta et al., 2019), land use (CoDyre et al., 2015; Gittleman et al., 2012), nutrient input (Small et al., 2019; Wielemaker et al., 2019) and other inputs such as compost and fertilizers (Csortan et al., 2020; Dobson et al., 2021; McDougall et al., 2019).
In-situ experiments	Experimental methods measure fluxes or conditions at farms and gardens.	Studies measure greenhouse gasses emitted directly from the soil or substrate, such as N_2O, which are known to be important in rural agriculture (Llorach-Massana et al., 2017b; Mendoza Beltran et al., 2022).
Survey of types of growing practices	Farmers/gardeners describe their practices in qualitative terms, i.e., 'what' they do, as opposed to quantitative measures of 'how much'	Surveys show the proportion of urban farms/gardens that make compost on-site, use organic growing methods, or apply pesticides (Kirkpatrick and Davison, 2018; Whittinghill and Sarr, 2021).
Ecosystem services	Indicators measured as part of a framework that seeks to measure the benefits that nature brings to humans.	Measurements of biodiversity of crops (Woods et al., 2016), inedible/ornamental plants (Cabral et al., 2017) and other organisms like bees (Quistberg et al., 2016); carbon sequestration (Cabral et al., 2017; Clinton et al., 2018); and stormwater runoff (Gittleman et al., 2017; Grard et al., 2018; Richards et al., 2015).
Material and energy flows	Materials and energy are tracked through systems as they are used or conserved.	Studies model insulation and avoided energy use from rooftop UA to buildings (Castleton et al., 2010; Clinton et al., 2018), reduced energy use from localized food distribution (Oliveira et al., 2021), and flows of waste and water in cities with more prevalent UA (Weidner and Yang, 2020).
Behavioral changes	Behaviors that are linked to environmental issues are measured for individuals before and after participating in UA.	Changes in sustainable consumption habits are measured, related to diets, waste treatment, and transport choices (Kim, 2017; Puigdueta et al., 2021).

Life cycle assessment	Environmental impacts are modeled for the inputs and outputs (direct and embodied) of a product or system	Life cycle assessments measure impacts such as climate change, water scarcity, and environmental pollution at the crop, farm, or city-scale (Dorr et al., 2021a).

Table I.3.1 A sample of possible methods to evaluate environmental dimensions of UA. The method is described, and examples of its application to UA are presented.

for LCA in 1997 (Heijungs and Guineé, 2012). This decade saw a great increase in interest and use of LCA, with a large increase of LCA studies in the relative share of the academic output (Zimek et al., 2019). Standardization increased the reliability and acceptance of LCA, which led to its use in legislation and policy during this period, although the method remained immature and had its problems (Guinée et al., 2011). The following decade of 2000-2010 was a decade of elaboration: a sort of renaissance period where LCA took off after the legitimization of standardization. Increasingly sophisticated problems were identified, and were addressed with a variety of new techniques relating to system boundaries, allocation methods, dynamic and spatially differentiated LCA, environmental input-output LCA, social LCA, and more (Guinée et al., 2011). Since 2010, the European Commission has released two detailed methodological frameworks on LCA, which provide more detailed information than the ISO standardization: the ILCD guidelines in 2010, which was effectively replaced by the PEF in 2018 (European Commission, 2010a, 2017).

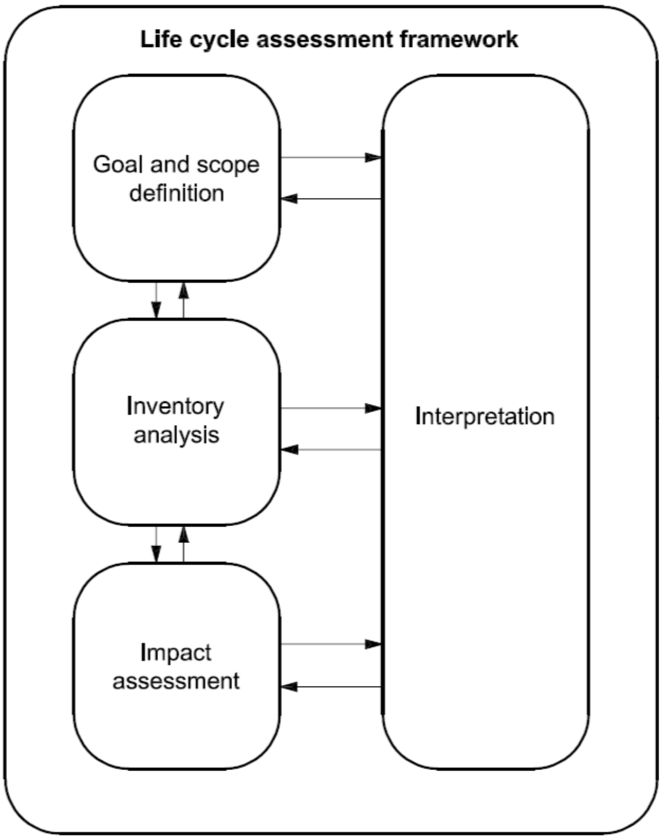

Figure I.3.1 The LCA framework, as defined by the ISO, includes the four iterative steps shown here.

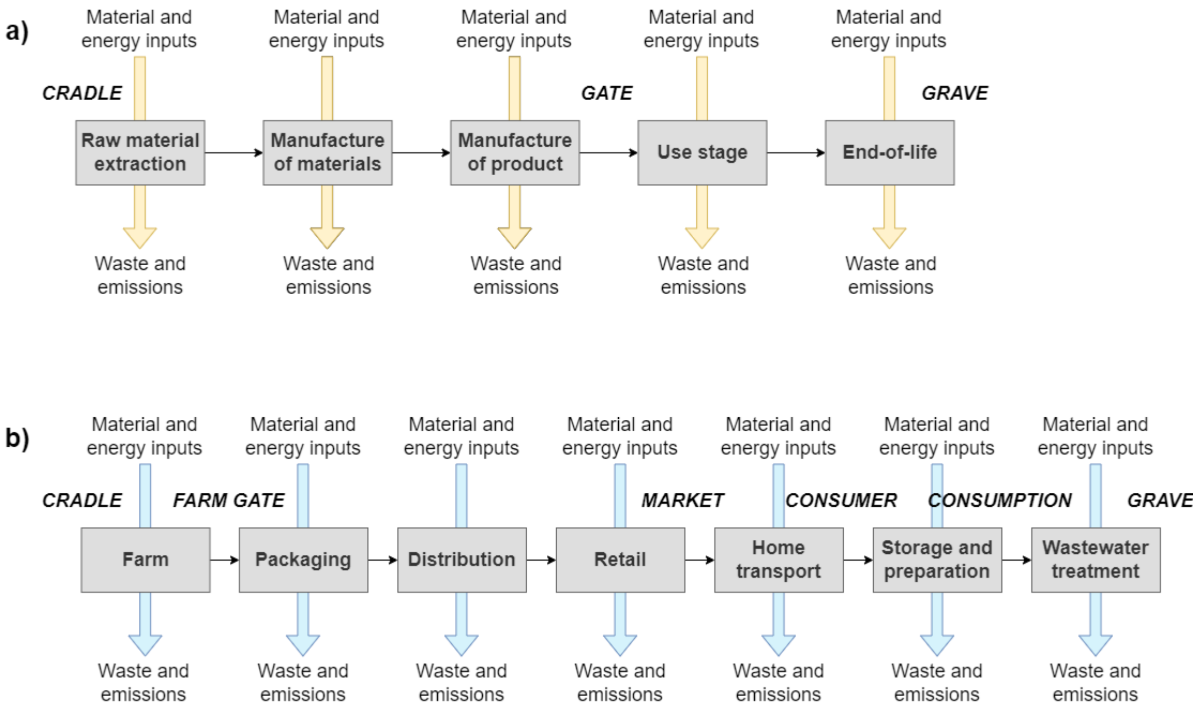

Figure I.3.2 The life cycle of a general product is shown in part a), starting with raw material extraction, and following through the life cycle to end of life waste treatment. In a life cycle assessment, environmental impacts of inputs and outputs (emissions) are modeled and summed for the entire life cycle of a product or system. Part b) shows a general life cycle diagram for a food product. Bold, italicized words represent the cutoff points for typical system boundaries, such as cradle-to-grave (a), or cradle-to-farm gate (b). Own work.

The LCA method involves four steps according to the ISO standards (ISO, 2006a) (Figure I.3.1). First, in the goal and scope definition step, the reason for carrying out the LCA is defined, plus the intended audience (Cucurachi et al., 2019). This is often accompanied by questions that the LCA aims to answer. Definition of the scope involves choosing the functional unit, system boundary, ways of dealing with allocation, impact assessment method, and databases, among other important decisions for setting up the LCA. The second step is the life cycle inventory, where flows of mass and energy as inputs and outputs of the system are defined and the data is collected. This includes the amount and type of flow. In the third step, impact assessment, the environmental effects of the physical flows defined in the inventory are modeled. Models used here are based on cause-effect models of natural systems, and can be performed using LCA software. Impacts can be calculated for separate impact categories, and/or for aggregated impacts using value-based weighing procedures. Finally, the interpretation stage involves identifying hotspots in the life cycle, evaluating sources of sensitivity, testing alternative scenarios, evaluating results in terms of the goal, and making recommendations and conclusions. LCA is an iterative process, and it is common to revisit and rescope stages throughout the assessment.

I.3.3. Life cycle assessment for agriculture

LCA was conceived and first developed for industrial products and packaging, and its application to agriculture appeared later (Andersson et al., 1994). The first conference dedicated to LCA of food and agriculture, the “International Conference on LCA in the Agri-Food Sector”, took place in 1996, and in 2007 it attracted 61 participants (INRAE, 2007).

Interest in the topic increased in the mid-2000s with the popularization of the "food miles" concept (Mclaren, 2010). Most recently in 2020 the "International Conference on LCA in the Agri-Food Sector" attracted more than 300 participants (INRAE, 2020). LCAs of rural agriculture and the conventional food system are now among the most common topics of published LCAs, alongside energy, infrastructure, transport, and waste (Zimek et al., 2019). Food and agricultural LCAs have been given special methodological attention due to the substantial differences between agriculture and industry, such as:

- **Product life cycle and scope**: An agricultural LCA can be expected to have different stages and scopes compared to a typical LCA. For example, a typical product LCA scope may use a scope of cradle-to-grave or cradle-to-cradle, which is less relevant for agricultural LCAs. A cradle-to-farm gate or to market/retail scope is more common for agriculture or food LCAs (Dijkman et al., 2018; Haas et al., 2000, p. 200).
- **Data availability:** Very few data were available regarding agriculture in LCA databases early on, although that is improving thanks to databases dedicated to the sector, such as AgriBalyse and AgriFootprint (Asselin-Balençon et al., 2020; Notarnicola et al., 2012; van Paassen et al., 2019). Still, data limitations are especially important for the fates of pollutants in the ecosystem, such as fertilizers and pesticides.
- **Site specificity:** Site-specific characteristics are especially variable for agriculture and can have a large influence on the results. These include gaseous emissions of N_2O, leaching of NO_3 to waterways, or dispersion of pesticides (Brentrup et al., 2000). This poses a problem for the nature of LCA, where one of its strengths is the possibility for site-independent and average assessments (Jolliet et al., 2015; Notarnicola et al., 2012).
- **Variability:** Inputs and production vary widely across farms based on climate, soil type, and farmer preference. These may vary between geographies, between farms in the same geography, or between years in a single farm (Mclaren, 2010). In contrast, industrial production for a given product typically has consistent processes.
- **Functional unit:** Food products do not necessarily share the same main function, so it is difficult to compare even similar foods. For example, although mass is commonly used as a functional unit, this excludes essential functions of food such as gustative quality, nutrient content, or protein content (Mclaren, 2010).
- **Multifunctionality:** Farms often produce multiple outputs, through crop rotations, or generation of low value by-products such as olive oil and wine pomace (Notarnicola et al., 2012). More broadly, crop production itself serves multiple functions, such as growing food, providing a livelihood, and managing land.
- **Impact categories:** Some environmental impact categories are especially important for agriculture, but are not treated thoroughly or consistently in LCA. These impacts are complex, dynamic, and difficult to model, and therefore are not as developed as more universally important impact categories. Some of these impact categories are described below.
    - <u>Land use</u> was originally evaluated using a sum of the land occupied (directly and embodied), although this fails to consider *impacts* on the land, following life cycle impact assessment practices (Andersson et al., 1994; Notarnicola et al., 2012). More sophisticated impact assessment methods have been developed by

combining indicators such as soil quality, carbon content, erosion, and microbial diversity, but there is no consensus on methods.

- Biodiversity impact assessment models for LCA have been explored for over 20 years, yet there is no complete and suitable method (Teixeira et al., 2016; Winter et al., 2017). Impact assessment models have focused on biodiversity impacts from land use, and impacts from other drivers such as climate change, pollution, and unsustainable use of land (such as monocultures) are much less developed (Winter et al., 2017). Experts recommend that a method should reflect the intensity of land use, the intrinsic conservation value/level of biodiversity in the location, and the vulnerability of biodiversity in the location due to existing pressures (Teixeira et al., 2016). Species richness is a common indicator, and is useful due to its relative simplicity, although its use is controversial since it excludes other scales of biodiversity, such as genetic and ecosystem biodiversity (Teixeira et al., 2016). Biodiversity is sometimes measured in separate assessments, parallel to LCAs. Even the UNEP SETAC Life Cycle Initiative suggests that “local biodiversity loss due to [land use/land use change] typically requires a level of spatial resolution and farm-specific knowledge for which other non-LCA tools may be more accurate” (Frischknecht et al., 2016).
- Water use, similar to land use, was typically considered using simple accounting of the freshwater withdrawals. This approach neglects impacts on water degradation and water scarcity. Many impact assessment methods have been proposed, and recent consensus building work from the UNEP-SETAC Life Cycle Initiative have recommended one method: the AWARE method (Boulay et al., 2018). This method models water scarcity, and is based on the “quantification of the relative available water remaining per area once the demand of humans and aquatic ecosystems has been met” (Boulay et al., 2018). Consistent use of this method is expected to make water scarcity impacts from agriculture LCAs more comparable.

These methodological issues have been extensively discussed in various guidelines and methodological development studies (Table I.3.2). Indeed, LCA research can usually be distinguished between case studies (applied LCA) and methodological studies (advancing knowledge in LCA methodology) (Klöpffer and Curran, 2014). The latter typically involve reviewing practices and choices in available LCAs, making recommendations for how to improve them, and/or identifying which methodological improvements need more research in order to become operational. Numerous methodological developments and reflections have been published for food and agriculture (Table I.2). These methodological developments have helped deepen and broaden the field of agriculture and food LCAs, and harmonization efforts have led to convergence between results, so that some generalizations can be made regarding environmental impacts of food. Such generalizations include trends that meat is highly impactful, especially beef, and making dietary changes to reduce meat consumption is typically the most effective way of reducing food impacts (Hallström et al., 2015; Poore and Nemecek, 2018). Heated greenhouse production is often more impactful than open-field production, even though it has higher yields (Bartzas et al., 2015; Clune et al., 2017). Organic agriculture is usually more impactful than conventional agriculture on a product basis, but less impactful on an area basis (although we should note that LCA does a poor job of characterizing several impact categories that are particularly important for organic agriculture) (Meier et al., 2015; van der Werf et al., 2020). The findings on the reduced food miles of local food and short food supply chains, however, have been much more complicated and context-dependent (Paciarotti and Torregiani, 2021).

Scope	Specific topic	Citation
Crop production		Adewale et al., 2018
Crop production	Organic	van der Werf et al., 2020
Crop production	Fruit orchards	Cerutti et al., 2014
Crop production	Vegetables	Perrin et al., 2014
Crop production	Climate smart agriculture	Acosta-Alba et al., 2019
Crop production	Microbial inoculants	Kløverpris et al., 2020
Crop production	Arable crops	Brentrup et al., 2004
Crop production	Soil carbon sequestration	Goglio et al., 2015
Crop production	Compost	Martínez-Blanco et al., 2013
Crop production	Crop rotation	Brankatschk and Finkbeiner, 2015
Agriculture and food	Harmonization	Audsley et al., 1997
Agriculture and food	Harmonization	Ponsioen and van der Werf, 2017
Agriculture and food	Ecosystem services	Tang et al., 2018
Agriculture and food	General	Caffrey and Veal, 2013
Agriculture and food	General	Dijkman et al., 2018
Agriculture and food	General	Mclaren, 2010
Agriculture and food	General	Notarnicola et al., 2017
Agriculture and food	General	Notarnicola et al., 2012
Food systems	Foods and diets	Cucurachi et al., 2019
Food systems	Diet inventory	Pernollet et al., 2018
Food systems	General	European Commission, 2017

Table I.3.2 Methodological reflections and frameworks for LCA are widely available for agriculture in general, and for specific agricultural sectors. A selection is presented here.

I.4. Why LCA for UA

For UA to have a legitimate role in alleviating problems from the food system and cities, it must be subjected to more critical system-wide assessments, quantifying both positive and negative impacts (Mok et al., 2013). LCA is a relevant method for such assessment. LCAs of

UA can identify which parts of UA systems contribute most to environmental impacts, guide eco-design for future UA projects, compare the environmental performance of different types of UA or types of UGI, compare the environmental performance of UA and the conventional food system, adequately frame the real benefits of UA, and provide critical evidence-based benchmark values that can be used for larger-scale assessments of urban food systems (Clinton et al., 2018; Santo et al., 2016). Numerous reviews of the benefits and impacts of UA conclude that LCAs are necessary to fill research gaps with evidence regarding the environmental performance of UA (Mok et al., 2013; Orsini et al., 2020; Petit-Boix et al., 2017; Santo et al., 2016; Weidner et al., 2019). Indeed, this is needed to test the resounding claims of benefits of UA, especially in light of critiques of UA. Such critiques suggest that farming practices may not be as ecologically sound as expected, with sometimes elevated water use, pesticide use, and widespread excess nutrient application; its small scale and fragmented nature may have inefficient resource use; and studies suggesting that some types of UA in certain contexts may have very large environmental impacts (notably climate change) (Goldstein et al., 2016b; Santo et al., 2016; Weidner et al., 2019). UA LCAs are necessary to bring nuance to the discussion around UA, and identify tradeoffs, points where UA achieves its claimed benefits, and points where it falls short.

LCA has only recently been applied to UA, and relatively few examples are available. No literature review or methodological reflection of UA LCAs had been performed at the outset of this project. Evidence suggested that the methods may be lacking: UA is a distinct activity; and LCA had to be adapted to applications of specific activities, especially agriculture. Furthermore, the environmental assessment tools used for rural agriculture are not adapted to the diverse, multifunctional, and urban nature of UA (Clerino and Fargue-Lelièvre, 2020; Feola et al., 2020). All of this suggested that it was necessary to take a step back and reevaluate the LCA method, in order to take a closer look at the environmental performance of UA.

I.5. Goal and research questions

The overarching goal of this thesis project was to provide and support deeper, broader, more rigorous and more consistent environmental performance assessment of diverse types of UA, using LCA. The specific goals were related to the methods and application of LCA and were twofold. The first goal was to propose a set of guidelines and recommendations for performing LCAs of UA that are holistic and well adapted to the unique characteristics of UA. The second goal was to apply these guidelines and investigate the environmental performance of diverse types of UA. Research to fulfill these goals was guided by three research questions:

1. How is UA distinct from other agricultural systems in ways that have implications for evaluating them with LCA?
2. What does LCA tell us about the environmental performance of UA?
3. How should we apply LCA to UA to make more consistent, complete and adapted assessments?

The original primary goal of the project was to develop a simplified LCA tool adapted for UA. This was rescoped early in the project, after finding that no methodological reflections or reviews on LCA of UA were available, and the topic was not mature enough to be simplified and streamlined. We went in the opposite direction: unpacking the literature, reflecting on the

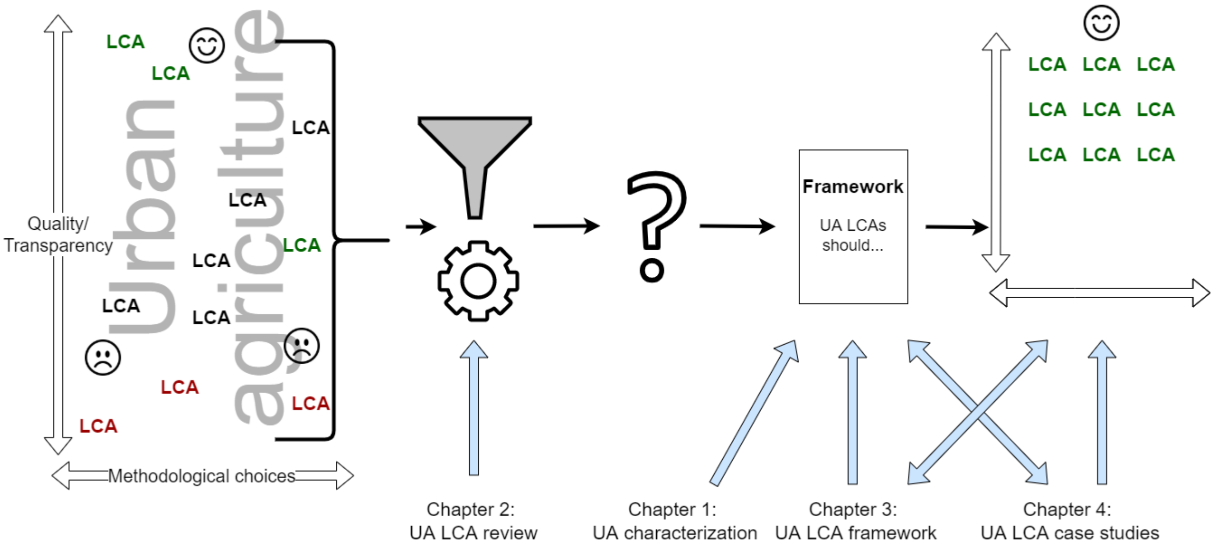

Figure I.5.1 The context and general methodology of the thesis project. Available UA LCAs have mixed levels of quality and methodological choices, resulting in a varied body of literature. In Chapter 2, we reviewed this literature, and faced difficulty in making sense of it and summarizing environmental impacts and benefits of UA. In response, we developed a methodological framework for performing LCAs of UA, which was based on the findings in Chapters 1 and 3 and featured in Chapter 4. An aspiration for this framework is that it leads to more consistent, high-quality UA LCAs, that will facilitate knowledge creation on the environmental performance of UA. This was demonstrated in our case studies in Chapter 3, which displayed lower variability in results than the variability found in the literature, despite them representing very diverse types of UA.

methods, exposing the many possible ways of doing these LCAs, and identifying how to improve the rigor and quality of UA LCAs. This rescoping led to the goal mentioned above: propose a set of guidelines and recommendations for performing LCAs of UA that are well adapted and holistic.

I.6. General methodology and outline of the manuscript

The general methodology of the thesis project is depicted in Figure I.5.1. The first main part of this research was an in-depth LCA of an urban farm case study, in order to learn through firsthand experience some of the difficulties and sensitivities of performing an UA LCA. The case study was an urban mushroom farm in the Paris area, and was chosen because it exemplifies some important but complex aspects of UA (circular economy, short supply chains, and more that we discovered along the way), and because the farmers were very engaged and had high-quality data. This is presented in Chapter 4, part 2.

After gaining some perspective with this experience, I took a step back and did an in-depth literature review, focusing on 1) history and development of LCA in general and LCA of specific sectors, 2) general methodological issues in LCA, 3) methodological issues in agricultural LCAs, and 4) UA LCA case studies. The review of UA LCA case studies was formalized and further developed, because no other in-depth review on this topic existed. The formalized review of UA LCAs is included in Chapter 2.

In parallel, I coordinated data collection and performed LCAs of eight urban farms and gardens in Paris, France and the Bay Area, California, USA. This started early in the project and overlapped with other work because most case studies required one full year of data collection, since they didn't already have records of past input use and production. The actual LCA modeling was done later in the project in parallel with the framework creation. This was an iterative process, where the issues and ideas that emerged from doing the LCA were included in the framework, and recommendations from the framework were tested and demonstrated in the case studies. The LCA of these case studies are presented in Chapter 4, part 1.

Also done in parallel was data collection led by other researchers at urban farms and gardens as part of the FEW-meter project (www.fewmeter.org). The project covered many facets of UA, but the data used in this thesis included measurements of food production and input use. I analyzed the data, interpreted results, and wrote them up. This provided a large-scale, evidence-based characterization of diverse types of UA (urban farms, community gardens, and individual gardens), allowing us to identify common points and make generalizations for the framework. This work is shown in Chapter 1.

All the lessons, ideas, challenges, and results from these steps culminated in the creation of the framework, covered in Chapter 3. This includes descriptions of challenges in UA LCAs, examples of how they have been addressed (in LCAs of UA or other topics), and recommendations for how to deal with them going forward. It also includes a (re)framing of goals for UA LCAs, and topics for future research.

The sections of this manuscript are not presented in chronological order, but rather in an order with a more logical flow of ideas. Chapter 1 presents a wide-ranging characterization of the food production and resource use of diverse types of UA, based on primary data collection with 72 farms and gardens in five countries. Chapter 2 features a literature review and meta-analysis of UA LCAs published before April 2021, which covers how the LCAs were done (focusing on their quality/consistency), and what they found (focusing on yield, water use, energy use, and climate change impacts). Chapter 3 is the framework for performing UA LCAs, and includes an overview of challenges, examples, and recommendations, and future research directions. Chapter 4 consists of two parts, both of which present original LCAs of urban farms and gardens. Part 1 is an LCA of eight urban farm/garden case studies, including four in Paris and four in the Bay Area, California. This focused on evaluating sources and magnitude of variability, which was identified to be particularly large among UA LCAs, and observing the differences between case studies in the two geographies. Part 2 depicts an urban mushroom farm next to Paris and focuses on the use of reused organic waste as an input and minimized transportation. Finally, Chapter 5 is a general discussion and interprets our findings in relation to the research questions, presents the contributions of this research, proposes future research directions, and provides final thoughts based on my personal impressions.

Chapter One: Large-scale characterization of UA

1 Chapter One: Large-scale characterization of UA

This chapter is composed of the original research article titled "Food production and resource of urban farms and gardens: a five-country survey", which is planned to be submitted to the journal Agronomy for Sustainable Development in April 2022. This research was done as part of the FEW-meter research project. Co-authors contributed to the conceptualization, methodology, review and editing, data collection, and PCA analysis. I contributed to conceptualization, all analyses and interpretation except for the PCA, data collection and curation, writing, reviewing, editing, and visualization.

Food production and resource use of urban farms and gardens: a five-country study

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Abstract

Urban agriculture (UA) is sprouting in many cities worldwide. Although food production is not always the main goal of UA, it is prevalent in all sites. For open-air urban farms and

gardens, the level of food production and resource use is poorly understood, and available evidence regarding its productivity and efficiency is lacking. This study aimed to assess the level of food production and the inputs used across 72 urban farms and gardens, representing 3 different types of UA (urban farms, collective and individual gardens), across five countries, using a participatory approach. Based on data from the 2019 growing season, we observed a large variability among sites regarding food production level, with high levels of crop diversity for some and clear differences in the use of space. We provided rare primary data on water, fertilizer, compost, and energy consumption. Our study provides a valuable contribution through a broad and quantitative primary dataset. Plus, we evaluated relationships in the data using comparative statistics and a primary component analysis (PCA), and identified trends (or surprising lack of trends), and clusters of similar gardens and farms. We here provide evidence of the extent that food production can be important, impactful, negligible or optimized across many contexts, which can inform the future development of UA.

1.1 Introduction

The environmental impacts of supplying food to cities are immense (Goldstein et al., 2017a). Urban agriculture (UA) is often promoted as means to alleviate these impacts and simultaneously provide multi-functional benefits towards health and well-being, among others (Gomez Villarino et al., 2021). UA is broadly defined as growing food in and around cities and interacts with the city through exchanges of materials, people, and values (Mougeot, 2000). While UA can take many physical forms, we focus here on soil-based gardens and farms that cultivate vegetables and fruits, as these are presently the most common forms of UA. An expected benefit of such systems is producing hyper-local, nutritious food for the city; nevertheless, growing food can consume substantial resources, including water, energy, land, and supplies such as fertilizers and pesticides (Campbell et al., 2017; FAO, 2011; Mohareb et al., 2017).

The level of food production output and the magnitude of inputs used in UA are poorly understood, partly due to its diversity, multifunctionality, and sometimes non-professional nature. Mixed evidence shows that UA can have both very large or small yields and can be resource efficient or inefficient, and explanations for this varied performance are scarce (CoDyre et al., 2015; McDougall et al., 2019). An accurate understanding of yields and farm inputs for UA is essential for evaluating its potential impacts on urban resource use and local food systems as the practice becomes increasingly popular. Such evaluations allow for more accurate projections of how much food consumed in a city can be provided by UA, or how much land it would require to feed a substantial part of the city (Grafius et al., 2020; Weidner et al., 2019). This would help clarify the effect of large-scale UA on the stocks and flows of material and energy that comprise a city's "metabolism" (Barles, 2009; Van Broekhoven and Vernay, 2018) In addition, increasing knowledge on yields and inputs is useful for evaluations such as life cycle assessments, that measure the environmental impacts of producing food in UA (Dorr et al., 2021a).

In order to study functioning UA in-situ—as opposed to research-oriented, ideally-managed urban farms—researchers take a citizen science approach, with urban farmers and gardeners collecting data about their practices (Pollard et al., 2018a). Many urban farmers and gardeners do not keep records about this information (Whittinghill and Sarr, 2021), and purchase and sales records are not always relevant or available for non-commercial and informal systems.

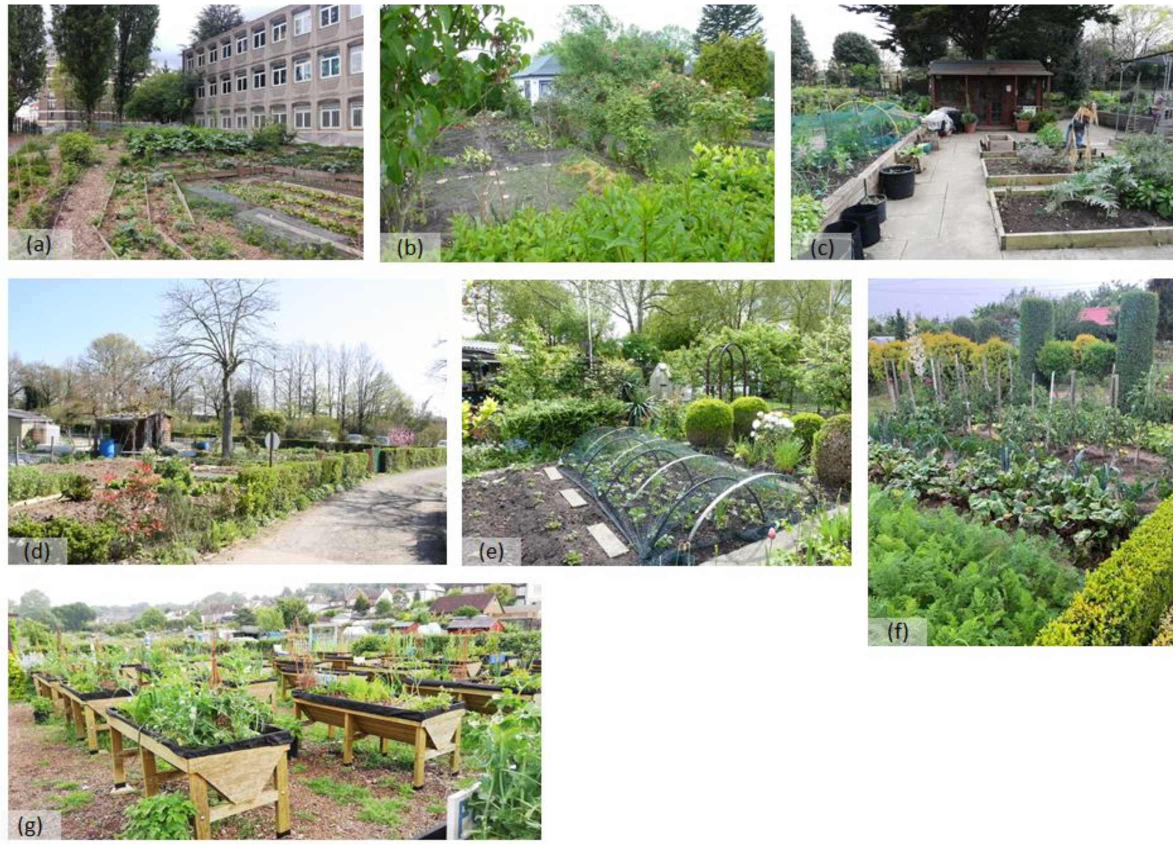

Figure 1.1 Illustration of study sites in the 5 countries. (a) Urban farms - 'Collège Pierre Mendès France', Paris, France. ©Baptiste Grard. (b) Individual garden – Bochum, Germany. ©Runrid Fox Kaemper. (c) Collective garden - London, UK. ©Silvio Caputo (d) Individual garden – 'Les Sorinières', Nantes, France. ©Louis Rosin. (e) Individual garden – Dortmund, Germany. ©Kathrin Specht. (f) Individual garden - Gorzów Wielkopolski, Poland. ©Lidia Ponizy. (g) Collective garden, Hertford, UK. ©Silvio Caputo.

Due to this lack of data, numerous studies evaluating UA use values from rural agriculture (Aragon et al., 2019; McClintock et al., 2013), or calculate and estimate values for yield and input use based on secondary data (Dalla Marta et al., 2019; Weidner and Yang, 2020). Studies that employ citizen science frequently characterize systems in a descriptive way, with surveys of what crops are grown and what practices are employed, but stop short of quantifying how much food is produced and how much of each input is used (Algert et al., 2014; Kirkpatrick and Davison, 2018; Woods et al., 2016). When such quantified data is collected, it is usually limited to a relatively small number of case studies (10-35), with one type of UA in one location (Algert et al., 2014; Csortan et al., 2020; McDougall et al., 2019; Pourias et al., 2015a; Sovová and Veen, 2020; Wielemaker et al., 2019). There are studies that have evaluated more than 50 case studies, but these usually have a rather narrow focus on food produced and do not include resources used (CoDyre et al., 2015; Edmondson et al., 2020; Nicholls et al., 2020). Dobson et al. (Dobson et al., 2021) had a large sample size (163 participants), and measured a suite of indicators covering food production and resource use, but only studied one type of UA: allotment gardens.

With this study, we helped fill this research gap by 1) measuring the level of food production and the inputs used in 72 urban farms/gardens representing three different types of UA, across five countries and 2) analyzing the patterns of food production and resources used. To achieve those goals, we measured mass and calories of food produced; a breakdown of yield per crop; and crop diversity. For resource use, we measured indicators including land use, water use and source, type and the amount of amendments such as compost and fertilizers, and energy use.

1.2 Material and methods

This study was carried out in the context of the FEW-meter project which aimed at understanding the impacts of UA on the urban Food-Energy-Water nexus (FEW) (www.few-meter.org). The full approach of the project and the methodology developed to measure the nexus is documented in Caputo et al. (2021). Data were collected in 2019 and 2020 in four European countries (France, Germany, Poland and the United Kingdom) and in the United States in a citizen-science approach (Ebitu et al., 2021). For this study we used data collected during the 2019 growing season. The research was divided in four steps: 1) Site selection, 2) data collection, 3) data processing and 4) data analysis. Each step is detailed below.

1.2.1 Site selection

Case studies were selected according to two main criteria: 1) farm or garden using soil or substrate (as opposed to hydroponic or other growing system using inert medium/substrate) and 2) willingness of farmer/gardener to be part of a citizen-science study. Data from 72 sites corresponding to three UA types were studied (Table 1):

- 9 urban farms, defined as productive spaces led by farmers with multiple goals (especially food production but also social and environmental functions) and that sell a portion or all of the food produced at the site.

- 8 collective gardens characterized by non-commercial purposes on land shared among gardeners.

- 55 individual gardens that were non-commercial with land divided into plots managed by individual gardeners. These included allotment plots and home gardens.

Cities had similar climate characteristics, but had variable populations and demographics (Table 1). Regarding climatic conditions, European cities are under the same climate zone (temperate oceanic climate, Cfb) while New York is under the humid subtropical climate zone (Cfa) (Beck et al., 2018).

1.2.1.1 *France*

In France, 16 sites were selected, including 11 individual gardens from an allotment garden association in Nantes and five urban farms (two in the Nantes area and three in the Paris area). Among the five urban farms, two are commercial farms in Nantes with the main goals of producing food. Two others are school gardens located in Paris, with the main function of education. The last urban farm in Paris focuses on professional integration and training as well as food production. The main goal of the allotment garden site is community cohesion and development.

All stakeholders were involved in the project thanks to the network of the French team—no financial incentive compensated their voluntary participation.

		Type of urban agriculture					**City demographics**		**City climate**		
		Total (n=72)	**Urban farm (n=9)**	**Collective garden (n=8)**	**Individual garden (n=55)**	**Years established**	**Population (2017)**	**Population density (2017)**	**Growing season duration (days) (2017)**	**Median temp. growing season (°C)**	**Annual precipitation**
Germany	Bochum	2			2	1947	364,920	2,510	178	14.4	841
	Dortmund	2			2	1938	586,600	2,087	178	14.3	698
	Lünen	1			1	1996	86,465	1,459	171	14.3	689
	Münster	6			6	1902-1922	313,559	1,030	171	14.3	652
France	Nantes area	13	2		11	1982-2016	309,346 (8,541)	4,745 (656)	225	15.8	787
	Paris area	3	3			2014-2018	2,187,526 (86,375)	20,755 (14,996)	304	16.7	672
Poland	Gorzów Wielkopolski	35	2		33	1975-2018	123,921	1,439	152	14.8	423
Great Britain	London area	5	2	3		2008-2020	8,825,000 (29,303)	5,575 (4,600)	208	14.4	655
United States of America	New York	5		5		2013-2018	8,622,698	10,636	226	18.3	1488

Table 1.1 Breakdown of case studies by location (city and country) and type of urban agriculture. Values in parentheses correspond to the neighboring, smaller city in the metropolitan area, where some case studies are located. Growing season duration is measured in the number of days between the last frost in the spring and the first frost in autumn. Climate data came from (NOAA, 2019), and demographics data came from (Eurostat, 2019; INSEE, 2018; U.S. Census Bureau, 2019).

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1.2.1.2 *Germany*

In Germany, 11 allotment plots were selected as case studies. They are located in the metropolitan Ruhr area (in the cities of Dortmund, Bochum and Lünen) and in Münster. The plots are used by individuals or families for both food production and leisure, and at least one third of the area must be used for the production of food according to German Allotment Law.

The participation of gardeners was organized with the help of the federal allotment garden association ('Landesverband der Kleingärtner Westafalen Lippe e.V.') who supported the project in recruiting participants by inviting their members to an informative workshop about the research project and the tasks ahead in November 2018. All participants signed a consent form declaring their willingness to contribute to the project over two years and received a small financial incentive afterwards (450€).

1.2.1.3 *Poland*

The 35 sites examined in Poland are located in Gorzów Wielkopolski, a city in northwestern Poland. Case studies were urban farms (2) and individual gardens (allotment plots and home gardens) (33). Enrollment for the project was carried out in 2018 with the help of the Polish Allotment Gardeners Association and the municipality. Gardener/farmer participation was voluntary, without any financial incentives.

All investigated sites are individually run. The main motivation for gardeners at individual gardens is recreation, but also food production for their own and their families' needs. The two urban farms are run individually, focused on sales at the local market.

1.2.1.4 *United Kingdom*

Case studies selected in the United Kingdom include two urban farms and three collective gardens. All case studies are in London metropolitan area.

The destination of food harvested varies across all case studies, with the urban farms and collective gardens selling their produce, and the latter donating food to their volunteers and gardeners. All case studies share social objectives and are connected with local groups and organizing activities to improve wellbeing, or to produce educational activities for local schools.

Sites were selected with the assistance of Social Farms & Gardens, a UK charitable organization that operates on behalf of community gardens, care farms and urban farms. The team launched a call to all SF&G London-based members, asking for expressions of interest to participate in the project. Researchers visited the 30 interested farms/gardens and partnered with nine sites; five sites collected data of sufficient quality to be included in the study. A small incentive was offered to each participant (£100).

1.2.1.5 *United States*

The US sites consist of six urban farms located within public housing developments in New York City. They are distributed across four of the city's five boroughs.

They are farmed by teams of young adults who are employed and supervised by Green City Force, a non-profit organization that provides workforce training and support to economically vulnerable youth living in public housing. Green City Force staff also provide technical support and labor for the farms. The project's goals are food production for distribution to public housing residents and ancillary services (e.g., educational tours, community events, cooking and nutrition instruction). Green City Force engaged the urban farmers in collecting operational data for the FEW-meter project together with CUNY researchers as a component of their farm training and to learn about the environmental and social impacts of urban agriculture.

1.2.2 Data collection

Three main methods were used to collect qualitative and quantitative data: observations by the research team during site visits, and monitoring food production and agronomic inputs.

1.2.2.1 *Site visits*

Data collection by the gardeners/farmers was accompanied by regular site visits by the researchers. These visits consisted of exchanging with gardeners/farmers about data collection and reliability, measuring spatial and infrastructure data, and observing the site and reporting significant changes in operation to favor data consistency. The frequency of visits depended on the site and the agreement with the stakeholder.

1.2.2.2 *Monitoring food production and agronomic inputs*

To record food production and inputs, we gave a harvest booklet to gardeners/farmers, with the exception of sites where a sufficient data collection system already existed (one site in France, two sites in the UK and all US sites). The gardeners/farmers collected data on (i) their food production including harvest date, crop types and names, quantity harvested in kilograms or other units (which were later converted to kilograms by researchers), and destination of the harvest; (ii) water used, in quantity and by water source (iii) energy use, recorded as the use of electricity and fuels; (iv) fertilizers, amendments and pesticides used (organic and synthetic); and (v) participation in social or educational events.

The layout of the harvest booklet differed slightly among study sites and was adapted to the individual needs of the gardeners/farmers. The research team provided intensive support for the task of entering data, regularly gave feedback, and provided materials such as scales for weighing produce and water meters when necessary.

1.2.3 Data processing

Data collected from March to October 2019 were included in the analysis. Data from 2020 were deemed unrepresentative and unreliable due to disruptions from the Covid-19 pandemic. All data collected were gathered in a cloud-based database using the Airtable software (*Airtable*, 2022). To analyze the data set, indicators representing food production and resource use were defined.

1.2.3.1 *Surface considered*

Three types of surfaces were considered to describe land allocation per site:

- Total site area: site area seen as the administrative limit of the project.
- Cultivated area: surface dedicated to cultivation of edible crops and other inedible plants or grassy spaces, including pathways within cultivated plots.
- Food production area: area dedicated only to cultivation of edible crops, as opposed to ornamentals, including pathways within the productive plots.

We then created two relative indicators:

$$Cultivated\ area\ (\%) = \frac{cultivated\ area\ (m^2)}{total\ site\ area\ (m^2)}$$

$$Food\ production\ area\ (\%) = \frac{food\ production\ area\ (m^2)}{cultivated\ area\ (m^2)}$$

1.2.3.2 *Food production*

Farmers/gardeners recorded harvest in mass or number of products (i.e. 1 head of lettuce). To express food production in a common unit, we converted all harvest records to mass with common conversions (e.g., 2.54 lb = 1 kg) or conversions supplied by farmers and gardeners (e.g., 1 bunch = 0.1 kg). Based on this measurement and food production area, the following indicators were calculated:

$$Yield\ (kg/m^2) = \frac{\sum harvest\ weight\ in\ fresh\ biomass}{food\ production\ area\ (m^2)}$$

$$Calories\ per\ m^2 = \frac{\sum calories\ per\ kilogram\ crop\ \times sum\ harvest\ per\ crop}{food\ production\ area\ \ (m^2)}$$

$$Economic\ value\ per\ m^2 = \frac{\sum local\ market\ price\ per\ kilogram\ crop\ \times sum\ harvest\ per\ crop}{food\ production\ area\ \ (m^2)}$$

Calories per kilogram of crop were determined using the USDA Food Intakes Converted to Retail Commodities Databases (FICRCD) and the USDA Food and Nutrient Database for Dietary Studies (FNDDS). When specialized crops were unavailable in one or the other database, proxies were used (e.g., basil for purple basil). Local market prices (i.e. at the city level) were collected at supermarkets and grocers near farms and gardens during the 2019 data collection period. We used proxies for crops that were not available locally (e.g., basil for purple basil).

To assess the cultivated diversity, the number of crops harvested was measured and a cropping diversity indicator was calculated with the following formula:

$$Crop\ diversity\ per\ m^2 = \frac{number\ of\ diferenct\ crops\ harvested}{food\ production\ area\ \ (m^2)}$$

1.2.3.3 *Resource use*

Fertilizer type and quantity were recorded at each application, and the nitrogen, phosphorus, and potassium (NPK) content for each synthetic fertilizer (percent by mass) was recorded. Using these numbers, masses of each synthetic fertilizer were converted to the NPK values with the following equation:

$$Synthetic\ NPK\ input = \sum \%\ NPK\ by\ mass\ per\ fertilizer\ \times mass\ fertilizer\ applied$$

Type and quantity of organic fertilizer and other supplies used (e.g., mulch, pesticide) were also recorded upon application but were not converted to NPK values.

For water consumption, a measurement system was set up in each study site either through automatic measurement (for instance using a water-meter) or by defining a counting system (conversion of irrigation time into volume for instance) Based on the total measured water use, we calculated water consumption per square meter of food cultivated area and per kilogram of fresh biomass.

Farmers and gardeners recorded their electricity and fuel use throughout the growing season. Electricity use was tracked using meters, and fuel was tracked using the volume of fuel consumed. We converted the volume of fuel to a common unit of kWh to be consistent with electricity using a conversion factor of 9.3 kWh/liter of fuel (U.S. Department of Energy, 2021). The energy used from both of these sources was summed to find the total energy use at the site.

1.2.4 Data analysis

First, data were cleaned and clarified through direct exchanges between researchers and farmers/gardeners. Eight of the initial 80 sites were excluded from further analysis due to data quality issues, leaving a final sample of 72 farms/gardens.

To explore potential explanations for the variation in results, we evaluated the relations between yield, water use, and several climate characteristics that were hypothesized to be relevant: annual precipitation, duration of the growing season (measured as the number of days between the last and first frost date in 2019), median temperature during the growing season, and average daily hours of sunlight. Because most of the variables had non-normal distributions, we used Spearman rank correlations. We also tested the correlation between several measured indicators, such as yield and water use, and yield and nutrient use. Relationships described in the text as significant had a p-value lower than 0.05. ANOVA and Tukey’s tests were done to evaluate the statistically significant differences between countries and types of UA. This was also used to test whether there were differences in yield and water use based on several technical factors including growing in substrate vs growing in the soil, and method of measuring water use. Analyses were conducted in R 4.0.5 software (R Core Team, 2020).

In addition, a principal component analysis (PCA) was done to analyze the variability in the dataset, and groups of similar study sites according to the main variables studied. To construct principal components, we used the following variables: percent cultivated area, percent food area, water use, yield, crop diversity, compost use and manure use (all in terms of m^2 food area). The PCA was conducted in R 4.1.1 software (R Core Team, 2020).

1.3 Results and Discussion

1.3.1 Level of food production

1.3.1.1 *Amount of food grown*

Yield varied from 0.2 to 6.6 kg/m^2, with a mean and median of 1.9 and 1.5 kg/m^2 (Figure 1.2a and 1.2b). The farm with the largest yield was substantially larger than the second largest (6.6 vs. 5.5), and was an urban farm in Nantes, France, which was the only case study where all of the production was done in a greenhouse. The mean and median value for calorie production per m^2 was 596 and 439 (Figure 1.2c and 1.2d). The maximum value was observed for an individual garden in France with 2,069 calories/m^2, while the minimum value was observed at an individual garden in Poland with 53 calories/m^2.

Differences were generally more important across countries than across types of UA: there were no statistically significant differences in yield or calorie output per m^2 between any UA

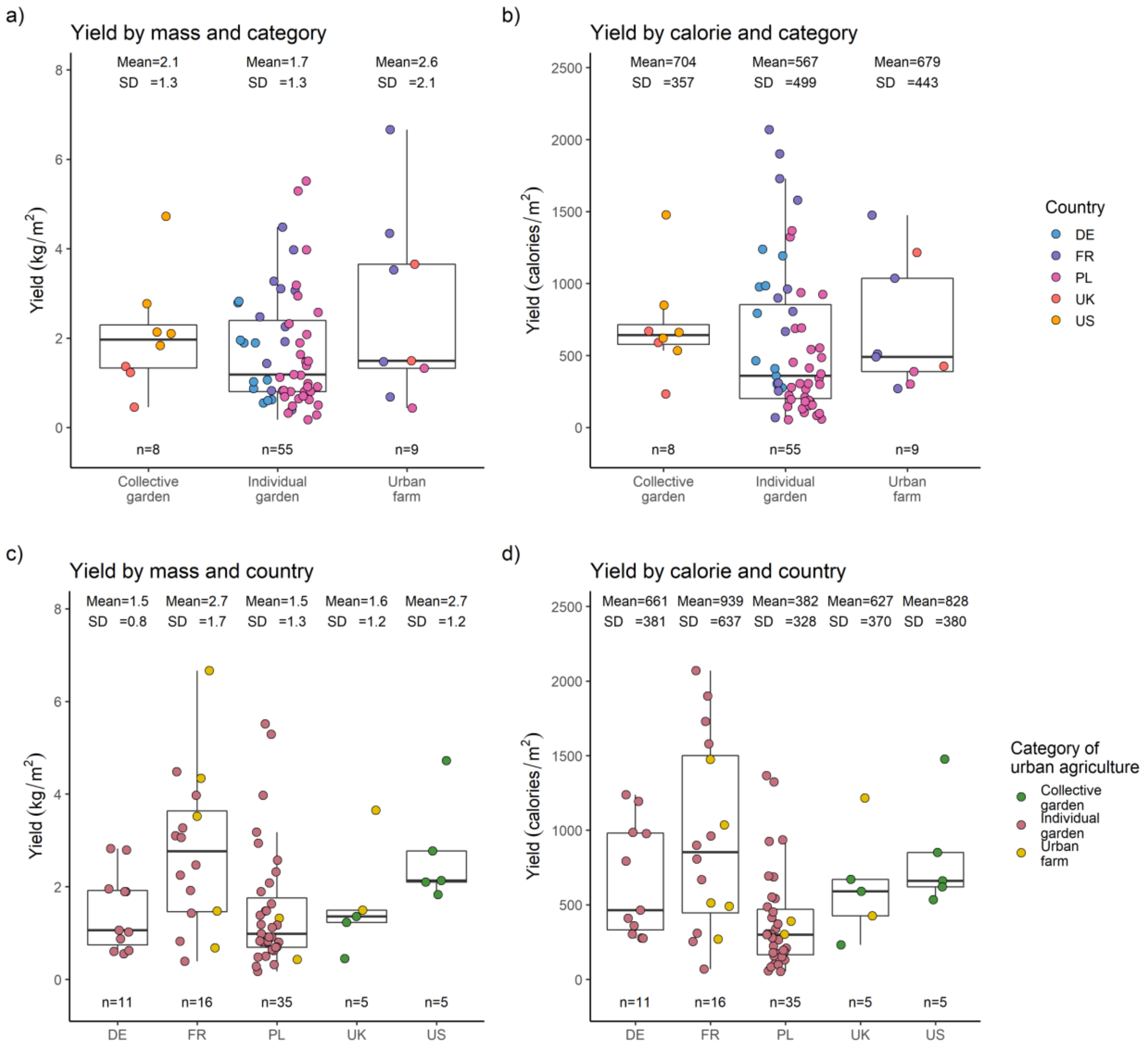

Figure 1.2 Food production expressed in yield by mass or calories per country or urban agriculture type. DE = Germany, FR = France, UK = United Kingdom, PL = Poland and US = the United States of America. Points represent results from individual farms.

types. The mean yield of urban farms was the highest, but there was a large standard deviation. Only Poland and France had statistically significantly different yields and calories produced per m^2. Farms in the US tended to have higher levels of production for all measures, along with French sites (although this was not statistically significant). Yield and water use by area had a moderate positive correlation (ρ= 0.52). Yield only weakly positively correlated with annual precipitation (ρ= 0.37), length of the growing season (ρ= 0.37), median temperature during the growing season (ρ = 0.31), crop diversity per m^2 (ρ = 0.39), manure use per m^2 (ρ = 0.25) and compost use per m^2 (ρ = 0.25). Results of correlation tests for the whole dataset are available in Table 6.1 in the Appendix. Grouped correlation tests were also done within each country and type of UA, and the results were similar (weak correlations with precipitation and length of growing season, no other correlations). Evaluating the effect of substrate types, we found slightly larger (but not statistically significant) average yields in farms/gardens that grew primarily on substrate or growing media (2.5±1.3 kg/m^2), rather than directly in the soil (1.8±1.5 kg/m^2). Other studies suggest that years of experience gardening and farming is positively correlated with yield (CoDyre et al., 2015), but we did not systematically collect this data. McDougall et al. (2019) also did not find an explanatory relationship between yield and several factors, including gardener experience, motivation for gardening, permaculture index, labor, and size of plot

We compared our results to data reported on yields from rural and UA sites. Weidner et al (2019) summarized data from FAOSTAT and reported average horticultural yields in developed countries ranging from 2.5-3.3 kg/m^2. Other studies suggest yields in community gardens, home and allotment gardens tending to reach 0.5-2 kg/m^2 (CoDyre et al., 2015; Dobson et al., 2021; Gittleman et al., 2012; Nicholls et al., 2020; Sanyé-Mengual et al., 2015b; Smith and Harrington, 2014; Sovová and Veen, 2020). Our results were similar to other UA results but slightly lower than rural horticulture averages, with most yields between 0.81-2.6 kg/m^2. It is difficult to generalize based on these case studies whether UA is particularly productive or not, but several sites demonstrated the potential for high production. Other studies have measured larger average yields of 3-6 kg/m^2 in community gardens and rooftop farming (Algert et al., 2014; Appolloni et al., 2021; McDougall et al., 2019; Pourias et al., 2015a) Similar to here, some studies report that individual cases achieve much higher yields than the average, reaching 10-16 kg/m^2 in allotment gardens, home gardens and rooftop farms (Boneta et al., 2019; Grard et al., 2020; Nicholls et al., 2020). Most professional and intensive UA reaches 5.4–7.1 kg/m^2, according to the review by Weidner and Yang (2020).

The harvest for each of the 25 crops with the largest harvest across all farms, which contributed to 90% of total food production, is shown in Figure 1.3, broken down by country. Tomato had the largest harvest by mass across all 72 farms/gardens, and spinach had the 25th largest harvest. Tomato appeared in the top 5 crops in each country, and lettuce was among the top 5 crops in all except for the case studies in the US. The importance of other crops varied across countries. Apples were the most important crop in German case studies, making up 18% of the harvest, but less than 2% of the harvest in other countries. Similarly, collard greens and kale were the most important in the US case studies, with 20% and 13% of the total harvest, but collard greens were not grown in any other country, and kale only had a minor importance in the UK (7% of the harvest) and was hardly grown in France (0.1% of total harvest). In terms of frequency, tomato was the most prevalent crop, and was grown at 93% of farms. This was followed by cucumber at 83% of farms, beet at 72%, green bean at 71%, carrot at 69%, and lettuce at 69%.

Analyzing the contribution of individual farms/gardens to the total production for each crop, we found that the tomato harvest in France was dominated by one farm that grew 75% of the tomatoes among the farms/gardens in that country, and lettuce production in Poland was dominated by a farm that grew 98% of all lettuce there. This was because three large farms—from France, the UK, and Poland—dominated the food production, accounting for 54% of the total harvest across all case studies. The harvest from these three farms was removed in Figure 1.3b to allow for better visualization of the breakdown of harvest from other farms.

Using mass of mixed crops is common in research quantifying food production in UA, as urban farms/gardens usually grow a variety of crops (Cameira et al., 2014; Kirkpatrick and Davison, 2018; Nicholls et al., 2020; Richards et al., 2015; Zainuddin and Mercer, 2014). This makes it difficult to compare the food production and resource use between case studies, as ultimately the products are different. Our case studies grew similar crops to other sites presented in the literature, to the degree that such similarity can be accurately described.

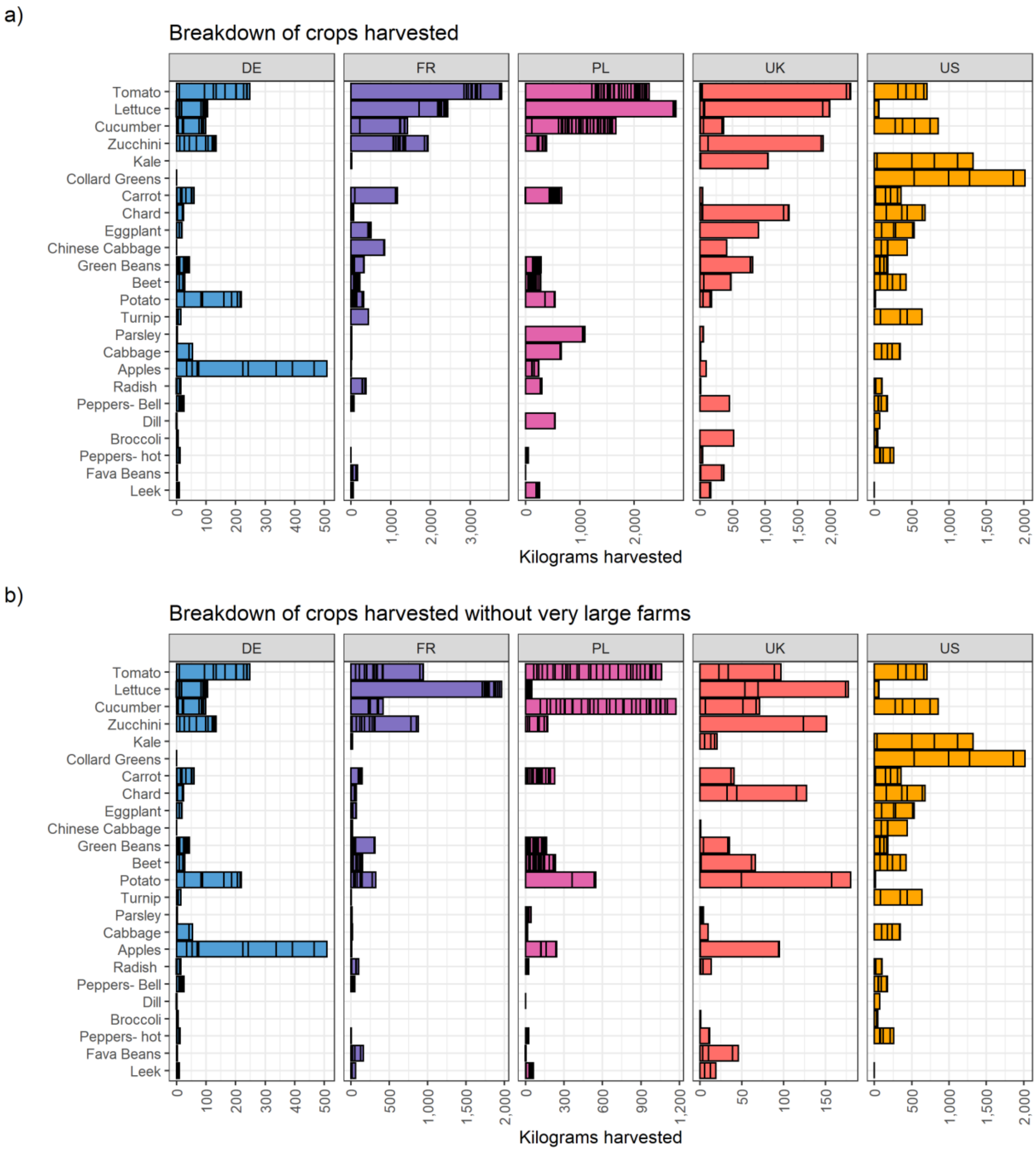

Figure 1.3 The breakdown of mass of crop harvested per country shown for the top 25 crops. This represents 90% of all harvest by kilogram. The black lines within bars represent harvest per farm, where large segments show farms with large harvests. In part a) results are shown for all farms. In part b), the three largest farms whose harvest dominated some crops were removed to improve visibility. Note that the values on the x-axis are different between part a) and b).

Nicholls et al. (2020) also found that tomato had the largest harvest (in 33 allotment plots in the UK), whereas legumes and beans were the second largest and were not particularly important here. Tomato, lettuce, and bean were the top three harvested crops across three years (in variable orders) across our case studies. The SEMOIR project that studied seven urban farms in Paris found that tomatoes represented 60% of mass harvested (Grard et al., 2022). Tomatoes and cucumbers had the largest harvests at home gardens studied in the US by Algert et al. (2014). Gregory et al. (Gregory et al., 2016) found that the most important crops at community gardens in New York were tomato, cucumber, bell peppers, and collard greens.

1.3.1.2 *Crop Diversity*

The number of different crops grown per farm/garden varied from 1 to 83, with 128 different crops recorded in the dataset. On average, 20±16 crops were grown per farm/garden, with a median of 16. Individual gardens cultivated a statistically significant lower number of different crops than other two types of UA. This was probably because these were smaller sites, or because individual gardeners may grow only what they want to support their individual diets, whereas other types of UA might choose a diversity of crops to appeal to customers and participants. There was no statistically significant difference in the number of crops grown at urban farms and collective gardens.

Because total number of crops grown may simply reflect the size of a farm/garden, we normalized this indicator by area. Crop diversity per m^2 showed the opposite trend: individual plots had higher crop diversity per m^2 (0.17±0.11) than collective gardens (0.07±0.05) and urban farms (0.07±0.09). There was no difference between the latter two. The allotment plots in France had especially high diversity (0.29±0.11), compared to allotment plots in Poland (0.12±0.08) and Germany (0.20±0.07). There were sharp differences in crop diversity per m^2 between countries, and were statistically significant between the US and Germany, the US and France, and Poland and France.

Twenty-three crops (18% of all crops grown) were only grown in one site. Only ten crops were grown at more than half of the farms/gardens. The frequency of all crops is in the Appendix. About half of the sites grew flowering plants, shrubs, or had native biodiversity areas, but it was beyond the scope of this research to collect detailed information about the non-productive areas and plants. Still, our findings on crop diversity suggest that these types of urban farms/gardens can be important sources of urban biodiversity. Other studies suggest similarly high, but variable, levels of crop diversity in UA, ranging usually ranged 5-43 (Kirkpatrick and Davison, 2018), 6-36 (Pourias et al., 2015a), 28-54 (Grard et al., 2022), and 18-70 (Gregory et al., 2016).

1.3.1.3 *Use of space*

Three urban farms had total areas that were much larger than the rest: in Paris, France, one site had a total surface of 16,690 m^2; in Gorzów Wielkopolski, Poland with a total surface of 12,390 m^2; and in Sutton (London area), UK, one site had a total surface of 28,726 m^2. Not considering these three sites, the farms/gardens had a mean total area of 773 ± 992 m^2, with a median of 400 m^2. There was extremely large variation in the total area, but 75% of farms/gardens had a total area between 295 and 600 m^2. Excluding the three exceptionally large sites, the largest farms were found in the US, with a mean of 3,065 m^2 (and were statistically larger than all other sites, which were statistically not different), followed by the UK with a mean of 1,288 m^2 (Figure 1.4a). This probably reflects the type of UA found in the US: most case studies were collective gardens, whereas other countries had many individual gardens, which were expected to be smaller. Differences could be observed according to UA types: individual gardens had an average total area of 434±528 m^2, compared to collective gardens with 2,415±1,577 m^2 and urban farms with 7,546±9,768 m^2 (or 1,685±561, excluding the three farms larger than 10,000 m^2).

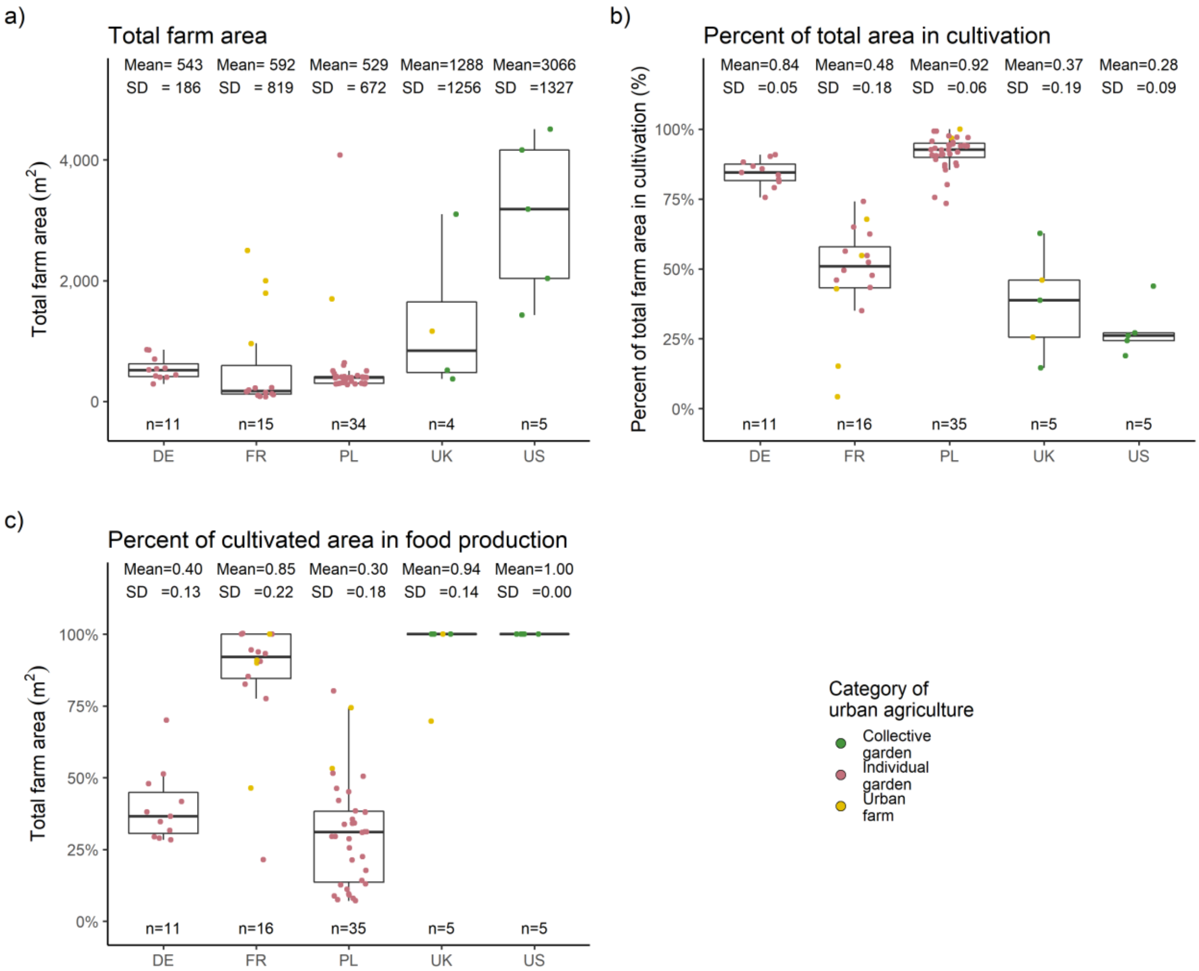

Figure 1.4 Use of space expressed per a) total farm size, b) percent of the total farm area in cultivation (including inedible plants and other green areas), and c) the percent of area in cultivation that was used for food production. This represents the breakdown of area for non-cultivation, other green spaces, and food production. In part a), three farms with very large areas were removed, and in parts b) and c) two of these three farms were excluded because they also had very large cultivated areas.

The large farms from Gorzów Wielkopolski, Poland and Sutton, England also had very large cultivated areas (11,987 and 13,200 m^2), but the large farm in Paris had a cultivated area similar to other sites. Excluding the two very large sites, the average cultivated area was 452±504 m^2, with a median of 358 m^2. As with total area, the sites in the US had the largest cultivated area, even though they had the lowest percent area in cultivation (28%±9%) (Figure 1.4b), meaning they dedicated larger amounts of area to non-cultivation uses like education or community meeting spaces. In contrast, individual gardens in Poland and Germany had the largest percent of areas dedicated to cultivation (91%±6% and 84%±5%, respectively). The average percent cultivated area compared to the total surface for all sites was 73%±26%.

Excluding the two farms with large cultivated areas, the average area in food production was 243±315 m^2, with the largest areas in the US (average of 799±291 m^2). The trend in percent cultivated area in food production was the reverse of the trend for percent cultivation: individual gardens in Germany and Poland had the lowest values, and the collective gardens and urban farms in the US and the UK had the largest values (Figure 1.4c). Nearly all of the cultivated land in sites in the UK, the US, and France was used for growing food, whereas less than half was used for growing food in Germany and Poland. Instead, gardeners allocated space to flower beds, hedges, lawn, and other biodiversity supports. This suggests the different potential between sites to either provide large amounts of food, or to provide space for biodiversity or aesthetic. Note that yield was calculated using the area in food production, so these differences in use of space are not reflected there. For both percent cultivated area and percent of food production area, the use of space varied more by country than by type of UA: almost all country-comparisons were statistically different, except for Poland and Germany, the UK and France, and the UK and the US. Percent cultivated area and percent food production area were the same for collective gardens and urban farms, and both were different from individual gardens. Other studies found average percent area in cultivation in UA ranging from 44-76% (Edmondson et al., 2020; Gregory et al., 2016; Pourias et al., 2015a). It should be noted that the definition of area under food production can vary between studies, with some considering only the productive areas without intra-parcel pathways (Grard et al., 2020). This must be accounted for when evaluating the food production potential of UA in cities.

1.3.2 Inputs and resource use

1.3.2.1 *Water management*

The most common source of water was municipal potable water, making up nearly all water used in 36% of farms. Other main water sources were groundwater wells in Münster (Germany), Gorzów Wielkopolski (Poland), and Nantes area (France). Collected rainwater was important in Bochum and Dortmund in Germany, where it accounted for more than 33% of all water used by gardens. In the UK and US, nearly all water used was municipal potable water, whereas a mix of several water types was used in Germany and Poland. Of the 43 farms/gardens that used potable municipal water, 8 did not pay for it (3 from France, 5 from the US). It is often hypothesized that UA could be positioned to use urban wastewater, but we found no examples of that among our case studies. Furthermore, rainwater collection was not very common, due to technical constraints (for instance the space necessary to store water or weight issues on rooftops), and the sanitary risk of sitting water. Due to water scarcity issues in many cities, and the impacts of tap water irrigation systems in life-cycle environmental impacts of UA (Dorr et al., 2021a), other methods of irrigation should be promoted in UA to improve its environmental performance. Despite this, other studies have found that municipal water is the main source for UA (Csortan et al., 2020; Whittinghill and Sarr, 2021).

One urban farm in France used an exceptionally large amount of water by area (2,802 L/m^2), due to leaks in the drip irrigation system and a poorly calibrated and excessive timed system. This value was excluded from analyses of water use by area. Most farms/gardens used 28.4-114 liters of water per m^2 of area in food production (Figure 6b and 6d). There was a large variation in the amount of water used, with a range of 1.7-1,312 L/m^2 and statistically significant differences between all types of UA, although there were no differences between countries (except for Poland and France).

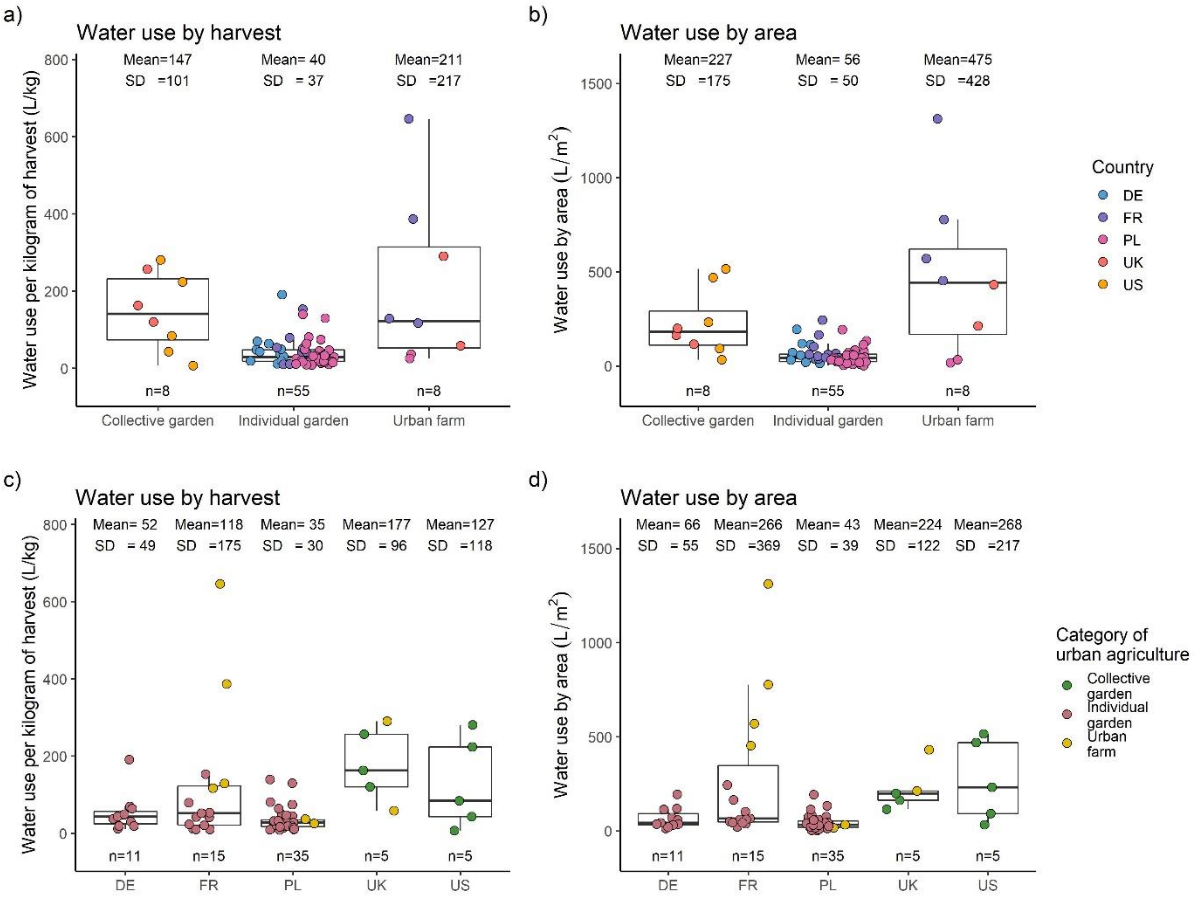

Figure 1.5 Water use expressed in terms of food produced (a and c) and of area in food production (b and d), and compared between types of UA (a and b) and countries (c and d). Water use from one French urban farm on an area basis was very large and removed from figures and analyses, and was 2,802 L/m^2. Similarly, on a food production basis, a different French urban farm had very large water use (1,942 L/kg) and was removed.

A different urban farm in France used an exceptionally large amount of water by kilogram of harvest (1,942 L/kg), and was excluded from analyses of water use by kilogram. There was a similarly large variation in water use based on food production, ranging from 6.9 to 646 L/kg of food, with a mean of 71.6 L/kg (Figure 6a and 6c). There were important differences in water use between types of farms/gardens, where individual gardens from all countries had substantially lower use than the other types. As a result, Germany and Poland (where nearly all sites were individual gardens) had the lowest water use, and sites in the UK and the US had average water use that was 3-6 times larger than that of Germany and Poland (although there was no statistically significant difference between any countries). France had both individual gardens and other types of UA, and had a mix of sites with high and low water use, but had a high average largely due to one site with large water use (in addition to the site with exceptionally large water use, which was excluded). France had both individual gardens and other types of UA, and had a mix of sites with high and low water use, but had a high average largely due to one site with large water use (in addition to the site with exceptionally large water use, which was excluded).

We explored several possible explanations for the variation in water use, and didn't find any strong factors, but identified some trends (and interesting lack of trends). We found a weak positive correlation with water use by area and precipitation ($\rho = 0.48$) and with yield ($\rho =$

0.52). Although it is counterintuitive that farms/gardens in cities with more rain use more water, we suspect that in cities with larger rainfall, crops that require more water may be planted, or farmers/gardeners may water excessively since water scarcity is not a concern. We found no trend between the amount of water used and the method of measuring water used: either calculations based on the number of buckets or continuous measurement with water meters. We evaluated the effect of the method of irrigation, and found that more water was used at farms using drip irrigation (average of 345±289 L/m^2) than those using hoses (104±201 L/m^2) or watering cans (74±63 L/m^2). This was surprising because drip irrigation is promoted as a precise, water-saving irrigation tool. Drip systems may use more water because they are prone to leaks, and they are often based on timers which imply standardized, consistent watering despite crop needs and weather. Comparing whether or not farms paid for the water used, we found that farms that didn't pay consumed more water than those that paid for some or all water (277±364 vs 74±76 L/m^2). Finally, systems growing primarily in soil had lower water use than those growing in substrate (99±207 L/m^2 vs 277±188 L/m^2).

Pollard et al. (2018a) pointed out that there are not many articles focusing on the measurement of water consumption in UA, and Whittinghill and Sarr (2021) found that water use was the least-recorded practice among urban farmers and gardeners surveyed. A survey of 163 allotment plots in the UK found an average of 16 L/kg, which was much lower than the average of 90 L/kg found here (Dobson et al., 2021). They also found extremely large variation in water use, with those on the higher end using very large amounts of water. Dorr et al. (2021a) summarized on-farm water use from 68 UA systems and found an average of 107 ± 121 L/kg. Ward et al. (2014) proposed that UA may use more water since polycultures are usually grown, and because different crops have different water needs, some are surely watered more than necessary.

1.3.2.2 *Fertilizer, amendments, and supplies use*

Substrate amendments and fertilizer were used by 94% of the farms/gardens. The most common input was compost, which was used at 52% of farms/gardens (Figure 1.6). However, this varied by country, with 100% of sites in the UK and the US using compost, and only 25% of sites in France. Only 8% of the individual gardens in Nantes used compost, while all urban farms the Paris area used it. Instead, the individual gardens in Nantes relied on manure, with 75% of gardens using it. Overall, manure was the second most common input and was used by 32% of all sites. It was mostly applied in individual gardens (91%), and in smaller and less dense cities such as Nantes, Gorzów Wielkopolski, Hertford, Bochum, Dortmund, Lünen, and Münster, and also more common among sites that grew directly in the ground than those growing in substrate (37% vs 8%). Sites growing in substrate were more likely than those growing in the soil to use compost (92% vs 44%) and potting soil (50% vs 9%). Nettle was the third most used supply, appearing in 29% of sites, followed by mineral fertilizer in 22% of sites (14% synthetic vs. 8% organic, such as calcium or rock flour). Again, this varied by country, with 64% of gardens in Germany and 26% of gardens in Poland using mineral fertilizer, and none among the other countries. All farms in the US used fish emulsion, which was characterized as "Other" in Figure 1.6.

Pesticides were not used as frequently as soil amendments: only 29% of farms used some type of pesticide. For 7% of farms this was an organic pesticide, and 24% of farms used a synthetic pesticide (and some used both organic and synthetic pesticides). These were mostly synthetic fungicides (used by 19% of farms), insecticides (11%), molluscicides (3%); organic

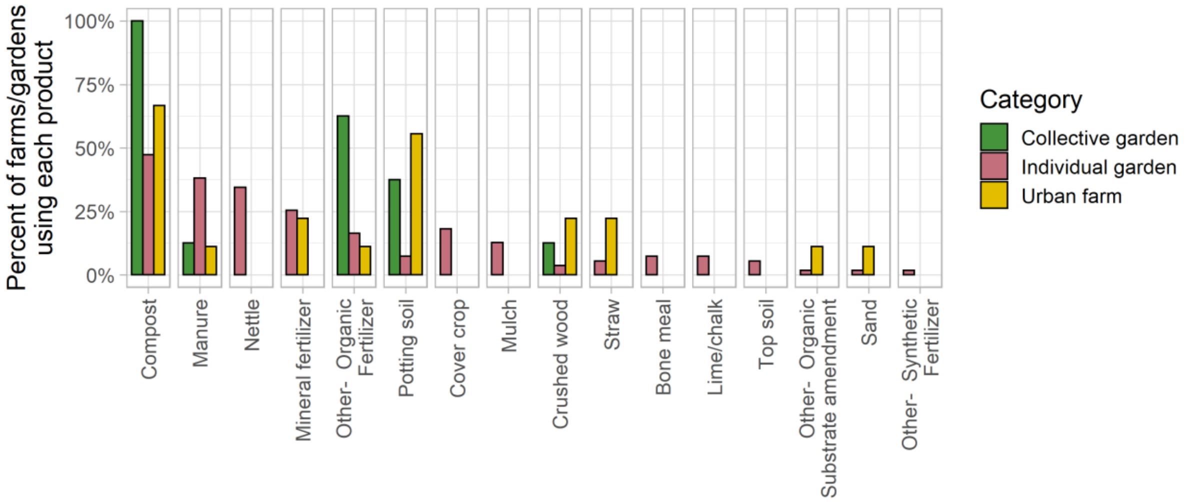

Figure 1.6 The frequency of use of various inputs is shown for the different types of urban agriculture.

molluscicides (3%) and other organic pesticides (6%). Frequency of pesticide use varied across countries: with 60% of farms/gardens in Poland, 38% in France, and 27% in Germany using pesticides. 51% of individual gardens, 22% of urban farms, and no collective gardens used pesticides.

Of the 38 farms/gardens that used compost, the mean, standard deviation, and median of liters of compost use per m^2 of food growing area was 9.6±11.6 L/m^2 and 4.85 L/m^2. The maximum amount was 43.7 L/m^2. The 20 farms that used manure used a smaller amount than compost, with a mean, standard deviation, and median of liters of manure use per m^2 of food growing area of 4.32±6.92 L/m^2 and 1.04, with a maximum use 25.7 L/m^2. Of the 12 farms that recorded use of synthetic mineral fertilizer, the mean, standard deviation and median of the grams of synthetic N, P, and K nutrient used was 0.0059±0.0060 g/kg and 0.0029 g/kg. As many farmers reported difficulty in measuring and recording the quantity of fertilizers and substrate amendments used, we assume that these values are an underestimate of the actual amount used. We observed that supply use was sometimes driven by shared practices among farmers/gardeners, or informal rules, rather than assessments of crop and soil needs.

Few studies have quantified the use of inputs such as fertilizers, compost, and other supplies in UA. They reported that inputs were difficult to track and have high uncertainty (Wielemaker et al., 2019). Most studies tracked the number of farms using a certain input and not the amount of that input used. Exceptions are Dosbon et al. (2021) who found an average of 1.9 L compost used/kg food in allotment plots, compared to an average of 5.5±6.3 L/kg here for all sites that used compost. Numerous surveys of UA have found that compost is the main input and practice, often followed by manure or other organic amendments, and that synthetic fertilizers and pesticides are less common (Dewaelheyns et al., 2013; Dobson et al., 2021; Edmondson et al., 2020; Gregory et al., 2016; Guitart et al., 2012; Kirkpatrick and Davison, 2018; Nicholls et al., 2020; Pollard et al., 2018b; Whittinghill and Sarr, 2021; Wielemaker et al., 2019).

1.3.2.3 *Energy use*

At 40% of farms/gardens, no energy was used on-site. The farms that used energy had a mean and median of 1.67±3.86 kWh/kg of food and 0.48 kWh/kg of food. Most of the sites had very small energy use, with 90% of values below 2.5 kWh/kg of food.

On-farm energy use is not usually measured for the type of urban farms/gardens studied here. This data is generally more available, and arguably more relevant, for indoor UA where lighting and temperature control may use substantial amounts of energy (Martin and Molin, 2019; Pennisi et al., 2019). Nevertheless, we included it here because large amounts of energy use are embedded indirectly throughout the food system: it is estimated that the food system accounts for 30% of global energy use (FAO, 2011). Much of this energy use can be expected to occur off-farm, such as in municipal water treatment, transportation, production of inputs but it is essential to quantify the on-farm use for a holistic view (Mohareb et al., 2017).

1.3.3 PCA

We performed a PCA to explore the relationships among our sites, and test the agronomic relevance of our three categories of UA. Indeed, the three categories were based on commonly used (but not strictly defined) typologies from the UA research community, and are mostly based on social and economic characteristics. Food production and resource use are rarely accounted for when making such typologies. As shown in Figure 1.7, the first two dimensions of the PCA explained almost 53% of the variability observed among the 7 variables included in the analysis. The variability of the first axis (32%) was mainly explained by the use of space (percentage of area under cultivation and food production), and the second axis was mainly linked to growing practices (manure used, crop diversity and compost used), while the third axis was associated with food production level (yield).

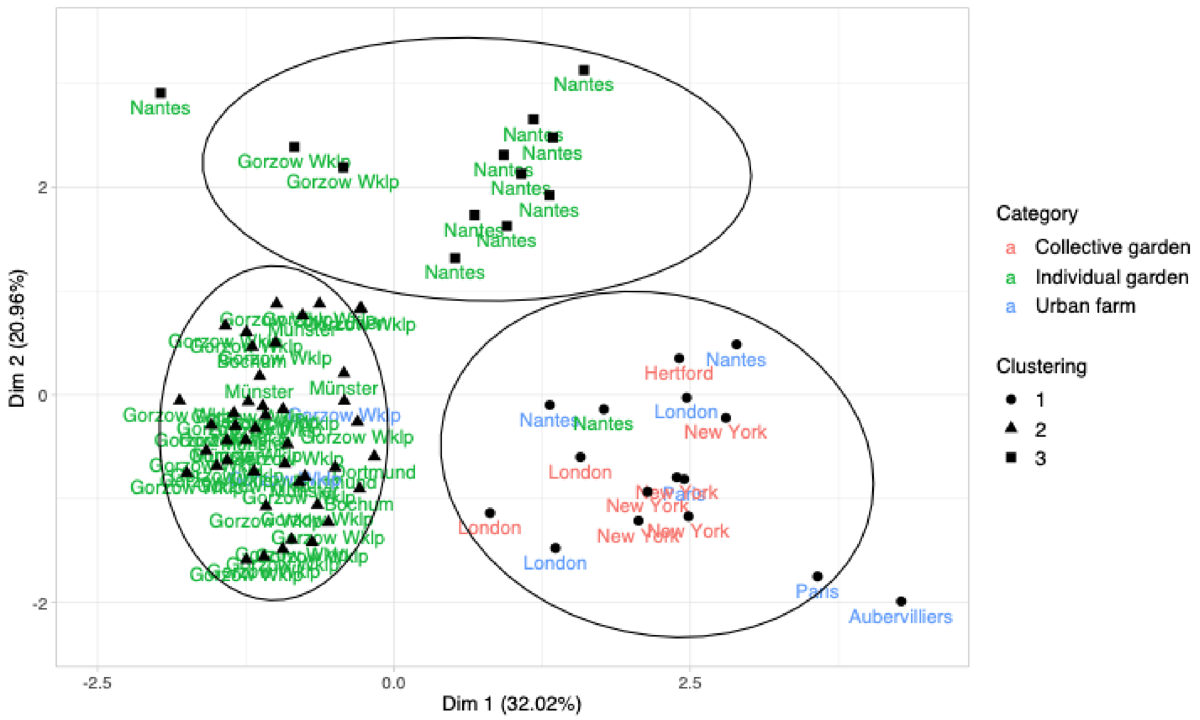

Figure 1.7: PCA analysis of the 72 study sites according to percent cultivated area, percent food area, water use, yield, crop diversity, compost use and manure use (all in terms of m^2 food area).

Clustering sites based on the PCA results showed that our categories of urban farm and collective garden were generally not distinct: most were placed together in cluster 1. The lower proportions of cultivated area and higher proportions of food production area in urban farms and collective gardens explained their cluster in comparison to individual gardens (see also Figure 1.4). Among individual gardens, ten allotment plots in France and two allotment plots in Poland were separated from the larger cluster of allotment plots, because they had lower proportions of cultivated area and higher proportions of food production area (similar to cluster 1), and used larger amounts of manure.

These results suggest that the commonly-used categories of UA we adopted differed mostly by use of space, and were not strongly related to food production and resource use. This casts doubt on the agronomic relevance of common UA typologies, implying that they should not be used for interpreting food production and resource use. For example, it seems intuitive that urban farms and collective gardens may differ in their food production and resource use intensity based on their different orientations, but our results suggest that this is not the case.

1.3.4 Limitations and outlook for future research

Most limits to this study were tied to the participative research approach. Because we could only include the sites that were willing to engage with us and collect the data, the case studies were not necessarily representative of UA in each city/country, and categories were not evenly split across countries. This is the reality of much case-based UA research (CoDyre et al., 2015). Also, some measurements had large uncertainties because they were self-reported. There were slight differences in our attempted systematic methodology of measurement, especially input use such as amount of compost and fertilizer, which farmers said was difficult to measure. Water use also came with large uncertainties, since farmers/gardeners reported difficulties with consistently reading and resetting water meters. Other studies highlight the difficulty in collecting accurate measures of water use in UA (McDougall et al., 2019; Pollard et al., 2018a).

Another limitation was that we only considered one growing season due to the COVID-19 pandemic, and did not assess temporal intra-site variability. Indeed, UA sites regularly experience large changes between years due to weather, changes in farm managers or gardeners, and changes in operation due to shifted objectives or other constraints.

The citizen science approach was time-consuming both for the gardeners/farmers who were asked to record their practices, and for researchers who dedicated large amounts of time to fill in missing data, convert between units or data types, and follow up with questions for farmers/gardeners. Indeed, urban farmers and gardeners do not typically collect data about their harvests and input use (Whittinghill and Sarr, 2021). We adapted data collection methods, allowing farmers/gardeners to submit data in a number of different units and types of measure, and obliging researchers to make sometimes complex conversions. For example, compost was measured using units of m^3, L, and kilograms; collected through tracking regular applications, evaluating purchase records, and measuring the amount of compost made; and assessed continuously or annually. This poses an issue for simplified tools aiming to help farmers/gardeners collect data, such as Farming concrete or the Harvest-ometer (Caputo et al., 2021), because they must be adapted to several different preferences and data input types.

Large research opportunities remain for documenting and explaining variations in food production and resource use of UA. We hypothesize that inconsistency in practices may play a role, where disruptions or changes in operation distort potential trends. Other factors not measured in our study may be more important for productivity and resource use, such as time dedicated to farming/gardening, level of experience/knowledge, level of professionalization, consistency of practices, who is mostly working in the farm/garden (i.e. one trained manager or children or punctual volunteers), crop choice, planting density, or a more complex assessment of motivations. Steps to improve studies such as ours include more accurate monitoring of resource use, inclusion of varied types of UA, and consideration of an even larger set of measurements, such as those mentioned above.

1.4 Conclusion

Our study demonstrates the feasibility and challenges of using a participatory approach to measure the food production and resource use of UA with a large sample size. A main contribution of this work was the creation of a dataset with high-quality primary data concerning food production, use of space, and crop diversity, across a large and diverse sample of UA (72 sites in five countries). We also quantified water, fertilizer, compost, pesticide and energy use, which was rarely done in the literature.

Though highly variable, food production for many sites was substantial. Some sites grew many types of crops, contributing to urban biodiversity. There were clear differences across sites in the use of space, with many sites using less than half of the total area for cultivation, or less than half of the cultivated area for food production. Water use was sometimes very large, and we identified higher water use in systems that didn’t pay for water, used drip irrigation, and were in rainier cities, indicating water mismanagement issues. Compost was the most frequently used input, and average quantities used were very large. Mineral fertilizers and synthetic pesticides were also used in many sites (mostly individual gardens). Our results contradict the common perception of UA as an intrinsically resource-efficient, innocuous activity. Still, our results must be interpreted with caution due to the high variability in results, as a consequence of the multifunctional and diverse nature of UA. Such high variability in our results was surprising, considering we applied a consistent protocol across all sites. This variability, unpredictability, and the unexpected low use of space for food production, raise doubts about the simulation of productivity and resource use at the city-scale, especially with secondary data and extrapolations. The common typology of UA that we used (urban farm, collective garden, and individual garden) had limited relevance for differentiating food production and resource use.

Food production and resource efficiency are universal in UA, regardless of farms’/gardens’ orientations. These must be better documented and understood to seriously account for UA in future urban food planning, and extrapolate the contributions and consumption of UA at the city-scale.

Chapter Two: How has UA LCA been done and what have they found

2 Chapter Two: How have life cycle assessments of urban agriculture been done, and what have they found?

This chapter is composed of the review article published in August 2021 in Environmental Research Letters, titled Environmental impacts and resource use of urban agriculture: a systematic review and meta–analysis. The only difference between this chapter and the published paper is the numbered references to figures and tables.

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 types, 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, plus 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.

2.1 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 (Artmann and Sartison, 2018; Feola et al., 2020; McEldowney, 2017). 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 (Alemu and Grebitus, 2020; Dobson et al., 2021; Goldstein et al., 2016a; Grard et al., 2018; Orsini, 2020; Specht et al., 2014). 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, 2020a). 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

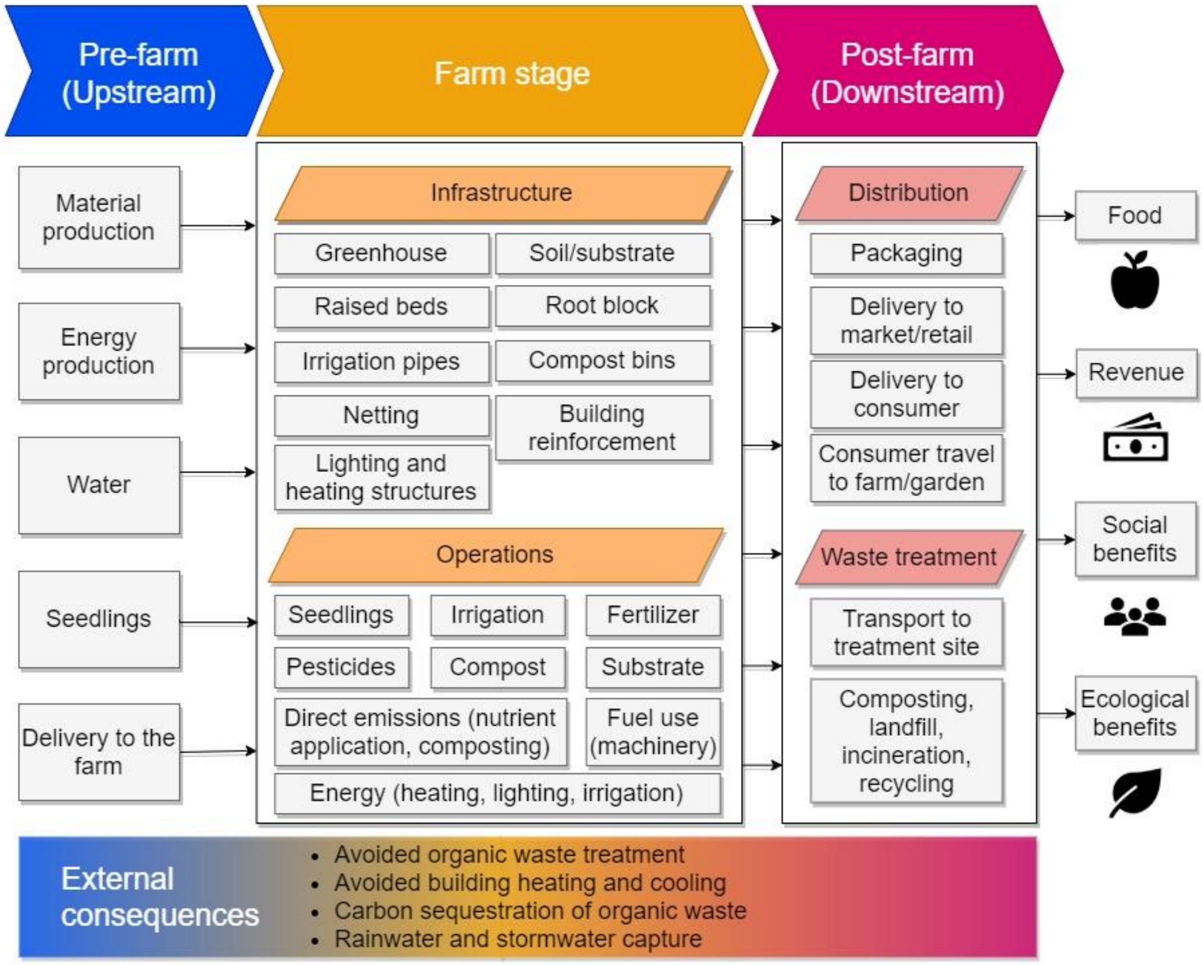

Figure 2.1 An example of what an UA LCA can be expected to include. This process diagram is an original figure created by the authors, based on informed opinion, designed to show an optimal UA LCA that includes the most important processes.

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, 2006a). 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.

2.1.1 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 2.1 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 kilograms of non–edible biomass are produced as a by– or waste–product (Boneta et al., 2019; Manríquez-Altamirano et al., 2020; Sanjuan-Delmás et al., 2018)). 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 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 provides insulation. These processes can have enormous effects on the final LCA results, or be relatively minor (Dorr et al., 2017; Goldstein et al., 2016b). 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.

2.1.2 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 (F 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 (Cleveland et al., 2017; Kulak et al., 2013). 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., 2020a) 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.

2.1.3 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:

1. What is the scope of UA LCAs in terms of types of systems assessed, crops and geographical areas?
2. What is the environmental performance and resource intensity of diverse forms of UA?
3. 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.

2.2 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).

2.2.1 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. The last literature search was performed in April 2021. To be included in the review, a study had to:

1. present an original LCA;
2. evaluate operations and agricultural production at an UA system or systems;
3. be a peer–reviewed journal article, scholarly book chapter, or peer–reviewed conference paper;
4. 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 one year; or avoided impacts per kilogram of food produced), were excluded (such as Sanyé-Mengual et al., 2018, 2017; 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 (Fisher, 2014; Perrin, 2013) on which a publication was based (Fisher and Karunanithi, 2014a; 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).

2.2.2 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, in a separate file in the online published version of this article (DOI: 10.1088/1748-9326/ac1a39). 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.

2.3 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.

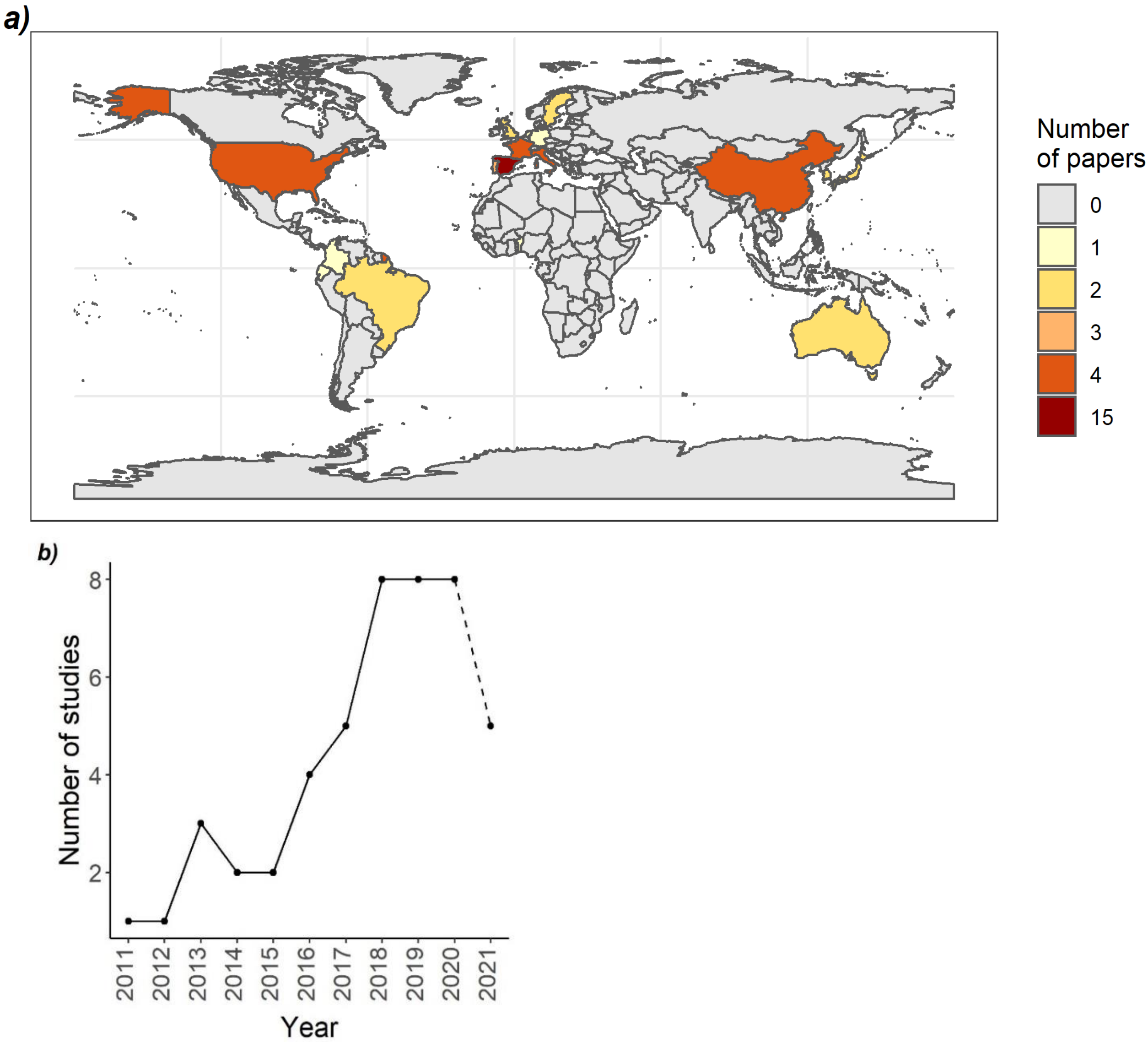

Figure 2.2 The map in part (a) shows that research performing LCAs of UA is largely centered in Europe, the United States, and China. Number of publications on the topic per year, shown in part (b), have increased over time, after first appearing a decade ago. This review only includes literature until April 2021, not the entire year, represented by the dashed line between 2020 and 2021

2.3.1 Systematic review

2.3.1.1 *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 2.2). This global distribution of studies between economically developed and developing countries has been identified in other LCA reviews (Clune et al., 2017; Laurent et al., 2014a; Poore and Nemecek, 2018), and likely reflects the prevalence of LCA application rather than UA interest, which has been widely studied in developing

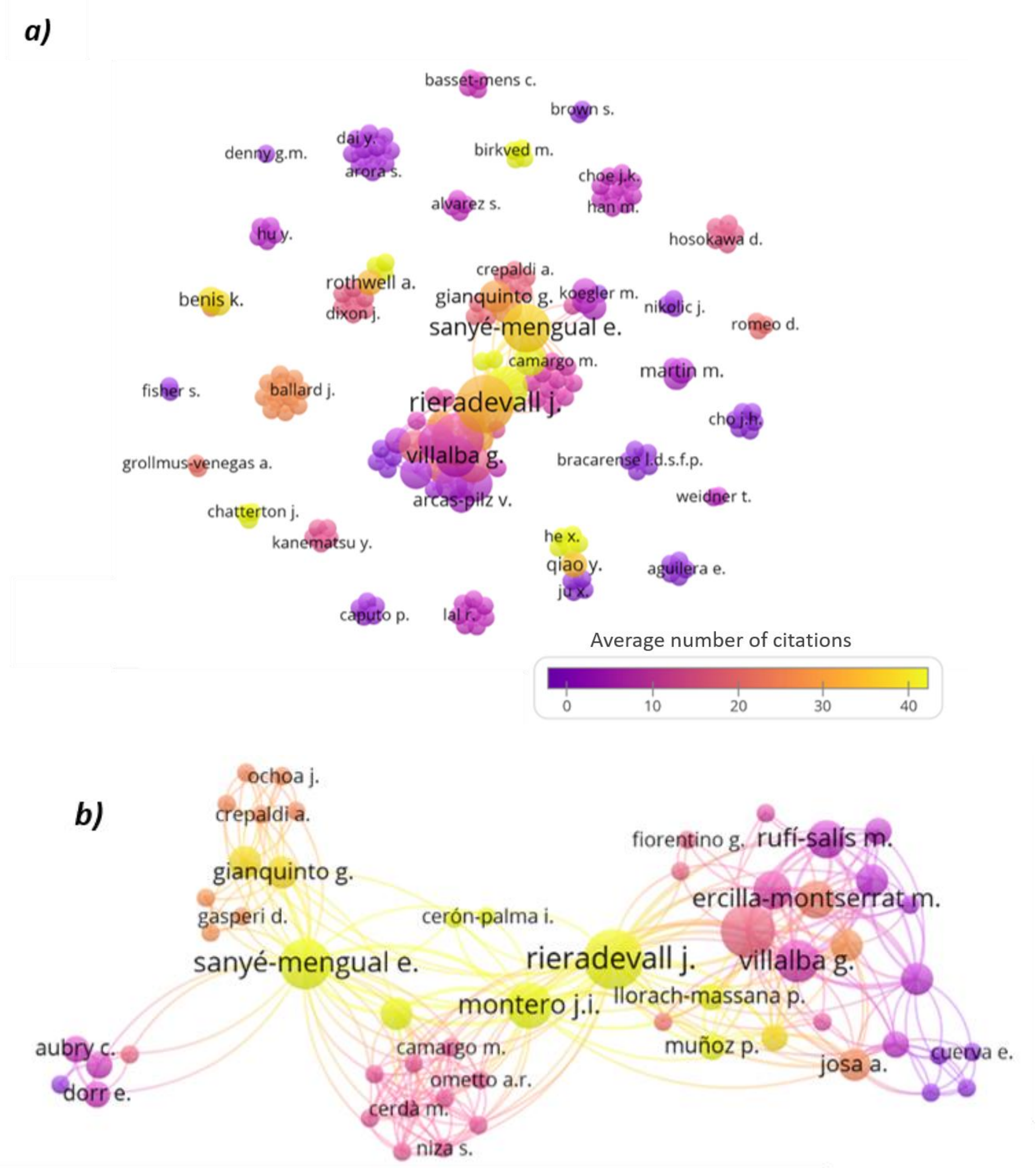

Figure 2.3 Co-authorship of papers in the review is shown, highlighting clusters of researchers. Size of circles relates to the number of publications each author has participated in, and the color scale indicates average number of citations for each author. Part (a) shows that there is one large cluster 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 integrated rooftop greenhouse at the Universitat Autònoma de Barcelona in Spain.

regions (Orsini et al., 2013; Zeeuw et al., 2011). 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 2.3a shows co–authorship through nearby placement of circles representing each author, which are more visible in Figure 2.3b, 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 10 papers (out of 47 total) and X. Gabarrell and E. Sanyé-Mengual each were co–authors in 8 and 7 papers. Many of these papers focus on an experimental integrated rooftop greenhouse 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 are 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 5 publications, and then Landscape and Urban Planning with 3 publications. Remaining journals had 2 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 8 publications annually (Figure 2.2b).

2.3.1.2 *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; Goldstein et al., 2016b; Romeo et al., 2018; Rothwell et al., 2016; Sanyé-Mengual et al., 2015b), or to confirm and quantify the reduction in impacts from UA (Benis and Ferrão, 2017; Cleveland et al., 2017; Kulak et al., 2013; 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 et al., 2020a). 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 (Dorr et al., 2017; He et al., 2016; Rufí-Salís et al., 2020a; Sanyé-Mengual et al., 2015b) or identify ways to improve the management of specific UA systems (Caputo et al., 2020; Liang et al., 2019; Rufí-Salís et al., 2020b).

2.3.1.3 *Types of farms and gardens studied*

The urban farms and gardens evaluated in the literature were highly diverse. Most papers studied IUA, but 8 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, rooftop greenhouses (RTGs) or integrated rooftop greenhouses (iRTGs).

Table 2.1 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

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 8 PUA papers, 75% evaluated commercial case studies, and 12.5% was 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 (Boneta et al., 2019; Caputo et al., 2020; Martinez et al., 2018). 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 2.1). 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 1,608 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 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 (Appolloni et al., 2021; Buehler and Junge, 2016; Pourias et al., 2015b). This crop choice is well reflected in the literature in this review.

2.3.1.4 *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 general 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).

2.3.2 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, 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 0.5-20 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. Cumulative energy demand 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 0.5-4.75 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 0.2-150 L kg^{-1} crop (75% of values).

2.3.2.1 *Yield*

We found yields for 77% of the production systems. Yields were calculated using the ground area, as opposed to the surface area or stacked, total vertical area for vertical 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 2.4a). 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

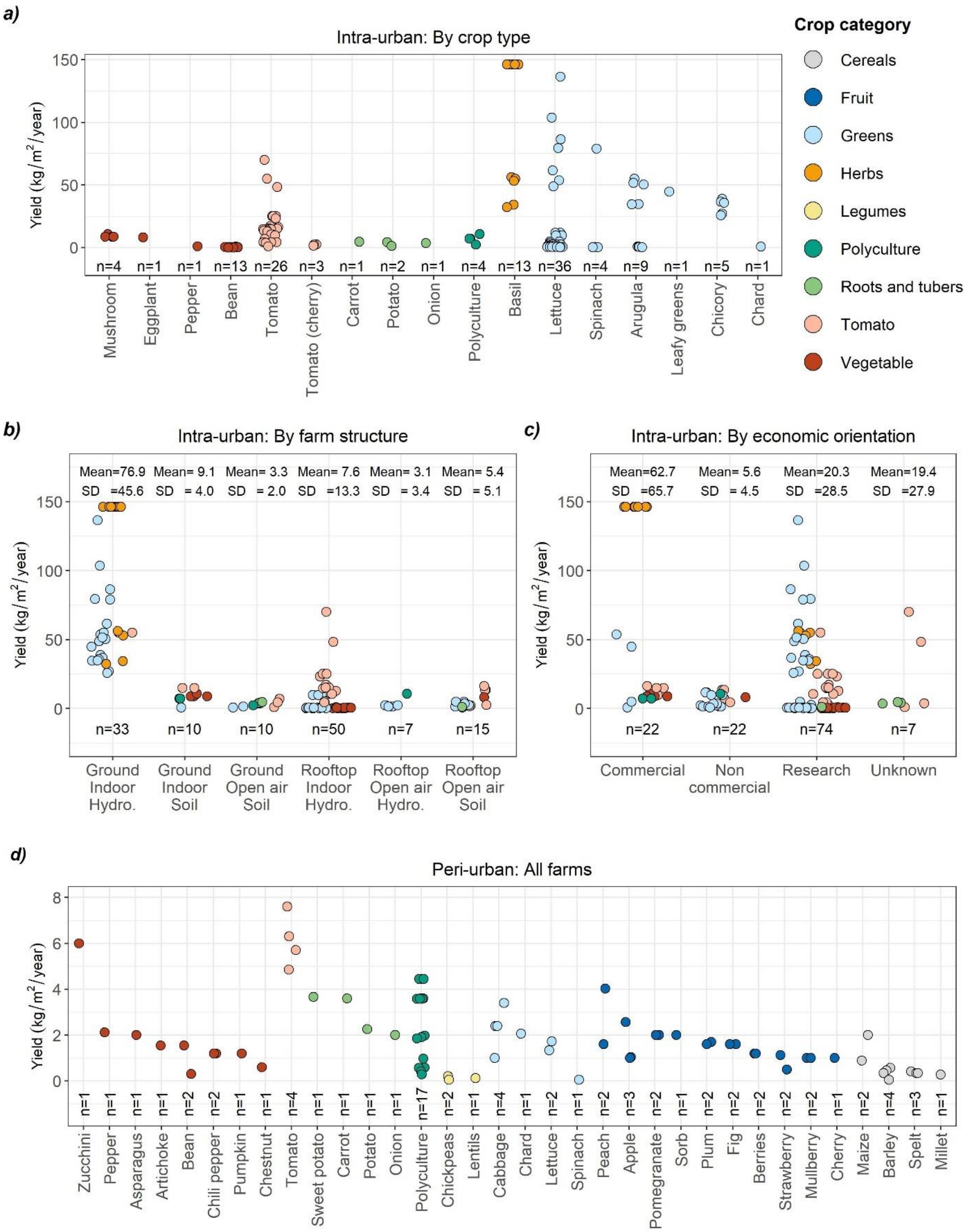

Figure 2.4 The yield results are shown in kilograms of crop grown per m^2 per year (fresh matter), for intra-urban agriculture by crop (a), by physical production system types (b), and by economic orientation (c). Part (d) shows results by crop for peri-urban agriculture. In (b) ‘Hydro.’ stands for hydroponics. Summary statistics for each crop are provided in the supplementary material.

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 2.4b). 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 (Arcas-Pilz et al., 2021; Rufí-Salís et al., 2020b, 2020a). 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 (Martin et al., 2019; Martin and Molin, 2019; Pennisi et al., 2019; Shiina et al., 2011). 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 2.4c), 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. 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.

2.3.2.2 *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 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

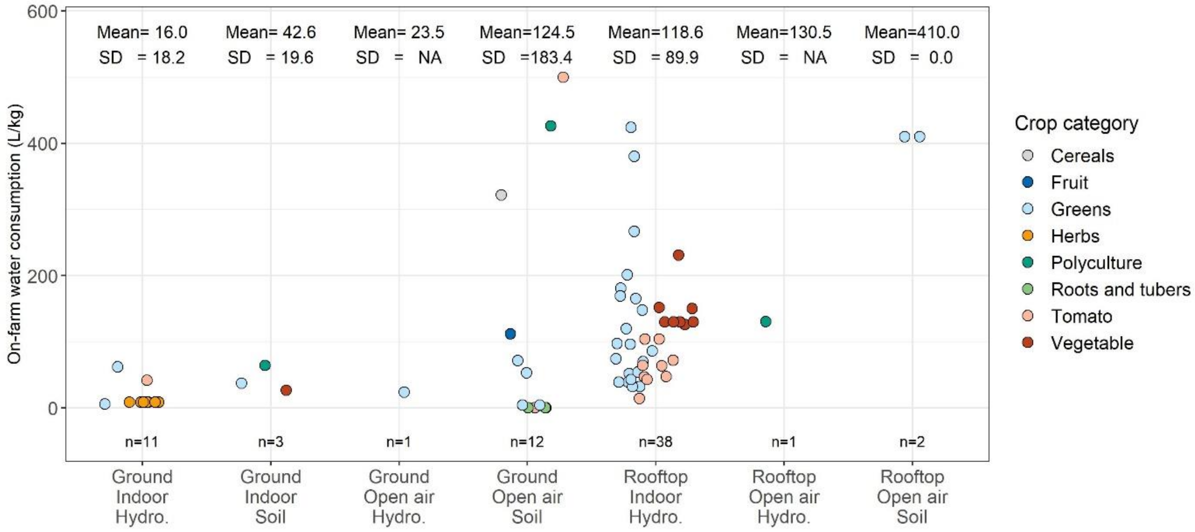

Figure 2.5 Water use for urban farms and gardens is shown for each physical system type. Note that this is on–farm water use (largely irrigation) and not life–cycle water use. Of the 68 results, 8 were from peri–urban agriculture.

types of systems except for ground–based open–air hydroponics (Figure 2.5). 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 2.5). 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 (Ledesma et al., 2020a; Liang et al., 2019; Perrin, 2013; Rufí-Salís et al., 2020a).

2.3.2.3 *Cumulative energy demand results*

There were results from 69 production systems for non–renewable cumulative energy demand (NR–CED) and 39 for total cumulative energy demand (CED), from 4 and 7 different papers, respectively (Figure 2.6). 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,

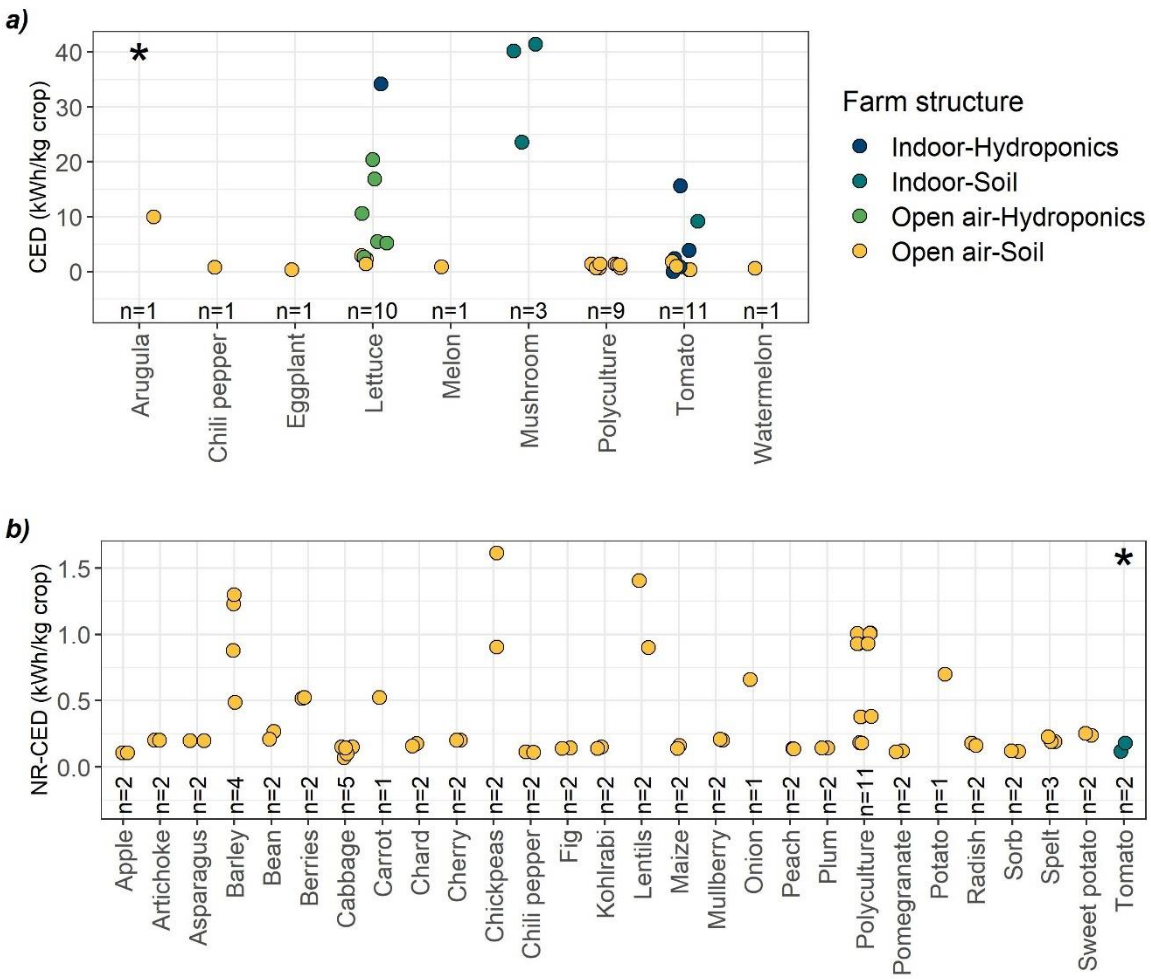

Figure 2.6 Part (a) shows total cumulative energy demand (CED) and part (b) shows non–renewable cumulative energy demand (NR–CED). Asterisks indicate where large outlier values have been excluded to improve the visibility of the figures— in part a) this was an indoor, soil–based system with a CED of 149 kWh/kg arugula, and for part b) this was an open–air, soil–based system with an NR–CED of 8.39 kWh/kg tomato. For CED, most values were from intra-urban agriculture, and for NR–CED nearly all values were from peri–urban agriculture from one paper (Caputo et al., 2020).

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 = 2.2e-16) and CED (r = 0.96, p–value = 2.2e-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 rooftop greenhouse 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.

2.3.2.4 *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 IUA systems was

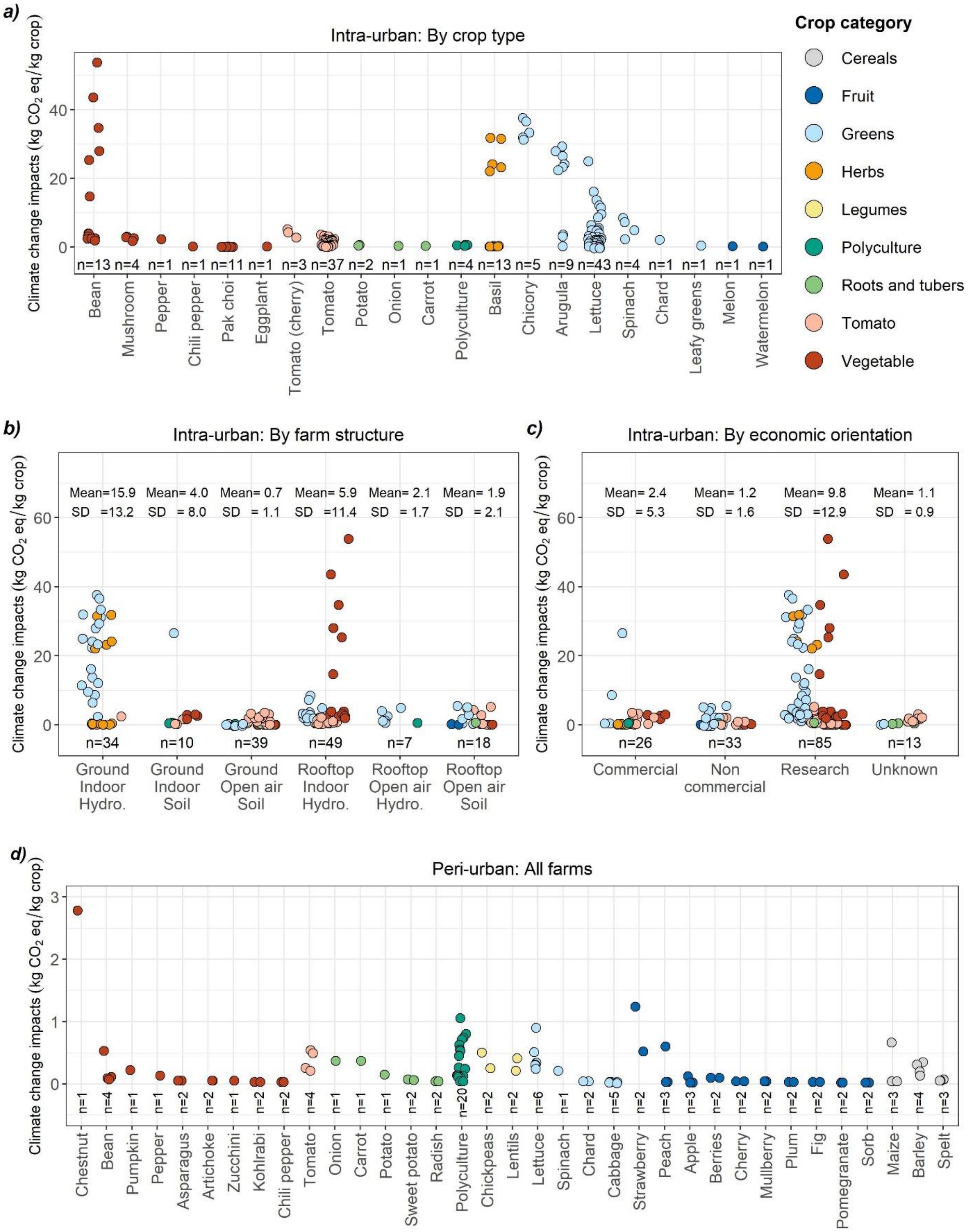

Figure 2.7 Climate change impacts are shown per kilogram of food produced per year, for intra–urban agriculture by crop (a), by physical production system types (b), and by economic orientation (c). Part (d) shows results by crop for peri–urban agriculture. One value of 6.1 kg CO2 eq/kg millet has been 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.

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 2.7a, 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 2.7b). The second largest impacts came from rooftop, indoor, hydroponics systems, where 88% of the results came from the same rooftop greenhouse 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 (rooftop greenhouses), 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 2.7c). Numerous systems used experimental production methods, including using biochar and struvite as inputs (Arcas-Pilz et al., 2021; Shen et al., 2020), 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 (Dorr et al., 2021b; Martin et al., 2019), which led to reduced yields, and may not be representative of how such systems would perform after research leads to improvements. Similarly, the integrated rooftop greenhouse 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 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 9 different papers, 5 different physical setup types, and 7 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, cumulative energy demand, 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 ± 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 6 hydroponics systems studied at the integrated rooftop greenhouse in Barcelona had impacts between 14.7-53.8 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 days, 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: 7 papers had systems with less than 0.1 kg CO_2 eq. kg^{-1} crop (5 from IUA, 2 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 one year (Figure 2.8). 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 ± 237 kg CO_2 eq. m^{-2} and 4.7 kg CO_2 eq. m^{-2}. Results were largely influenced by 12 systems from 4 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) (Goldstein et al., 2016b; Kikuchi et al., 2018; Pennisi et al., 2019; Shiina et al., 2011). These systems also had large average yields (70 ± 30 kg m^{-2}) and food–based climate change impacts (12 ± 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 2.8 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,

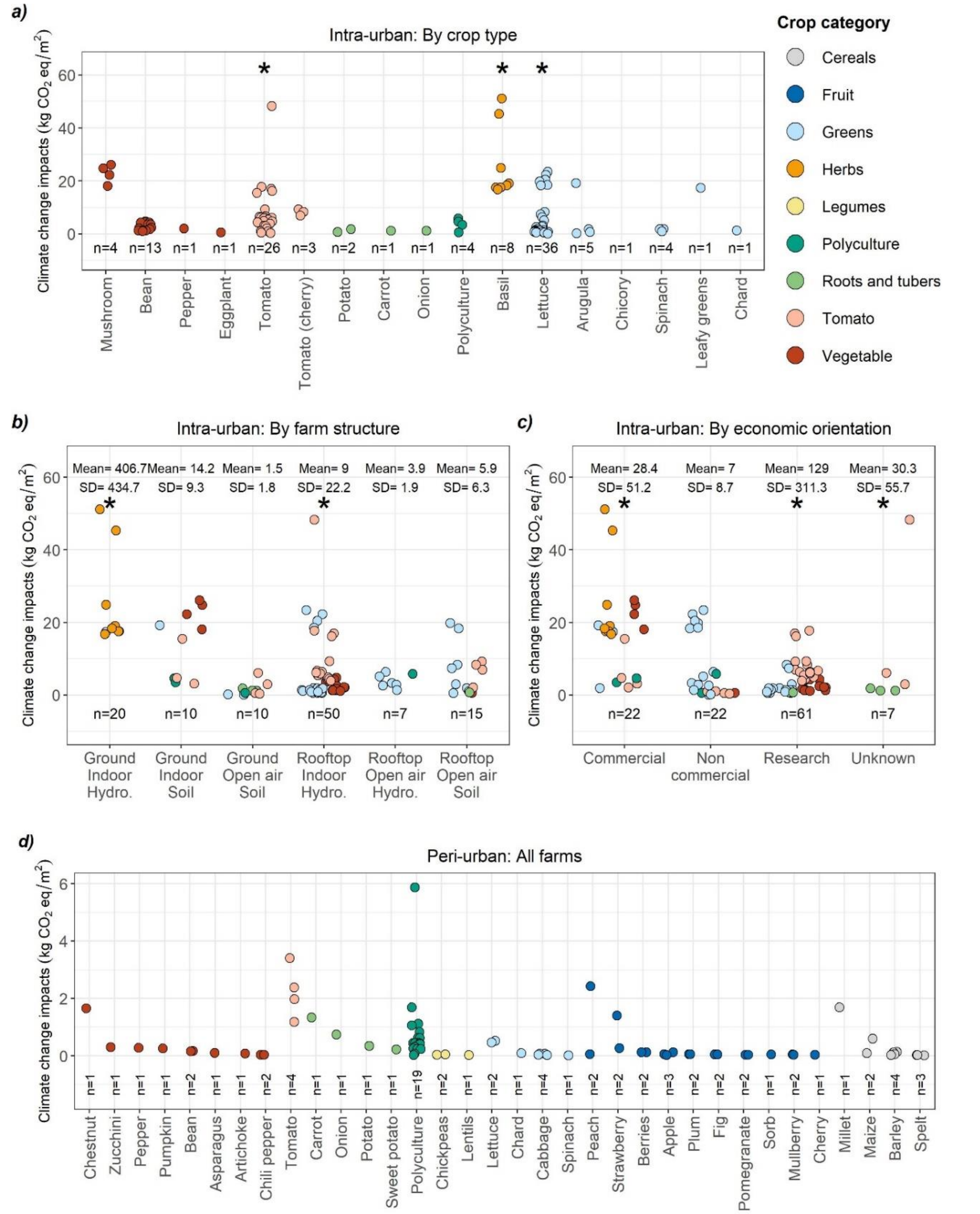

Figure 2.8 Climate change impacts per m² land occupied per year are summarized here, for intra–urban agriculture by crop (a), by physical production system types (b), and by economic orientation (c). Part (d) shows results by crop for peri–urban agriculture. In (b) 'Hydro.' stands for hydroponics. Note that 12 outlier points have been excluded, with values of 130-985 kg CO_2 eq/m², 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 4 different papers. These values have, however, been included in calculation of the mean, standard deviation and number of observations. Asterisks the groups where values have been excluded.

PUA had lower impacts than IUA, with a mean of 0.51 ± 0.90 kg CO_2 eq. m^{-2} and 0.14 kg CO_2 eq. m^{-2}.

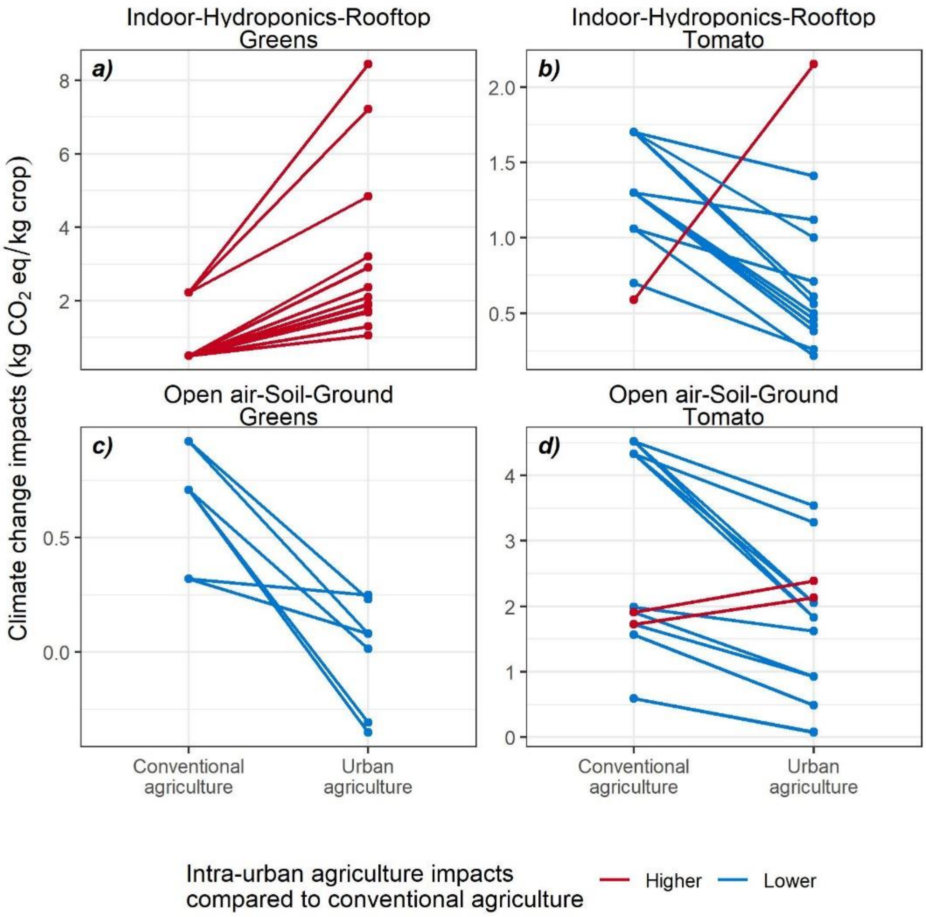

Figure 2.9 In many papers in this review, climate change impacts of the studied urban agriculture systems were compared to local, specific, conventional agriculture systems, for the same crop and physical farm/garden type. We summarized those comparisons here. Red lines represent comparisons where urban agriculture 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.

2.3.3 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 crop from IUA were lower than the conventional system in 41 out of 68 comparisons (60%), and higher in 40% of comparisons (Figure 2.9). 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 2.9a). Open–air, soil–based IUA systems and IUA tomatoes usually performed better than conventional agriculture (Figure 2.9b–d).

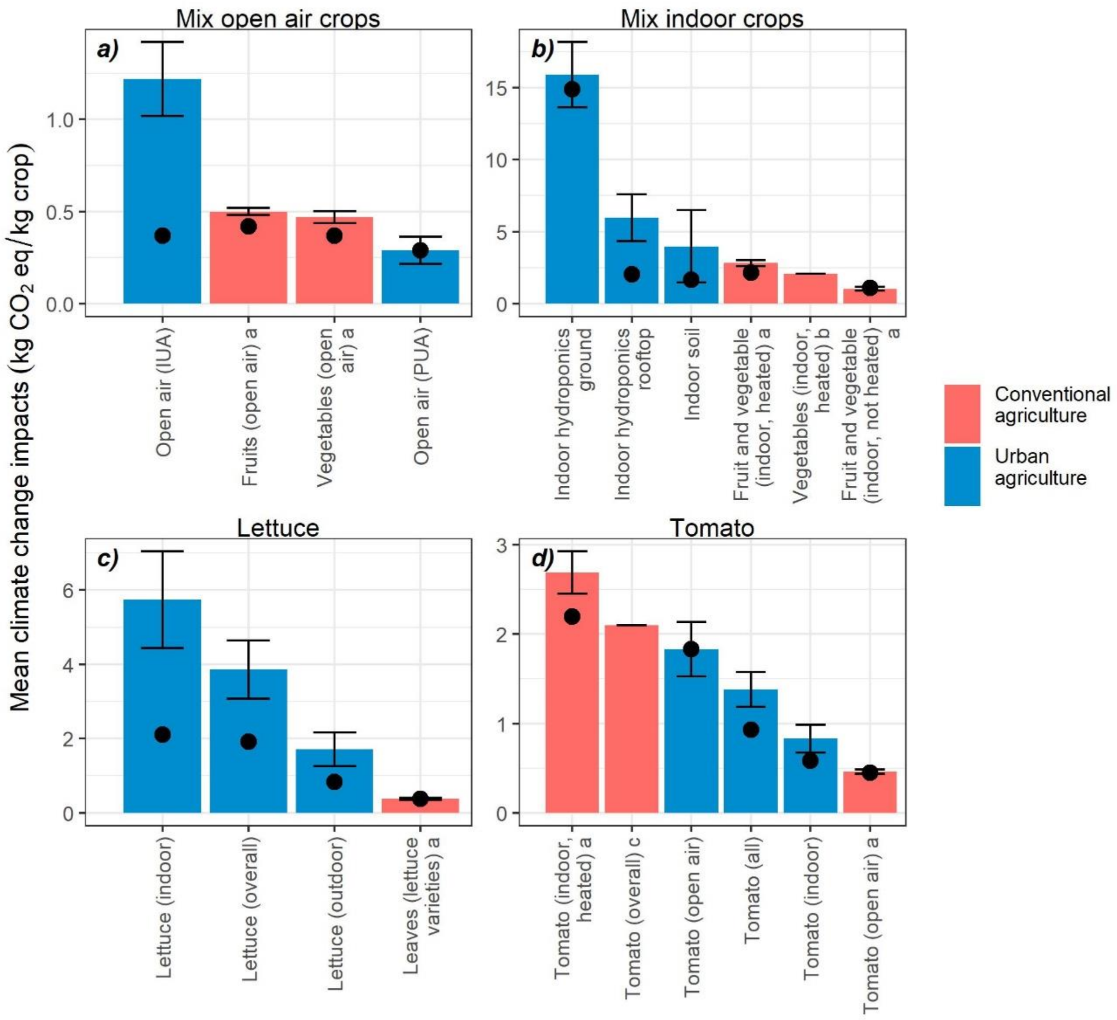

Figure 2.10 Climate change impacts were compared for urban agriculture systems (calculated in this review) and conventional agriculture systems from other food LCA review papers. The bars show the mean value of each group, with standard error bars, and the points show the median. Note that the y–axis is different for the different plots. In part a), IUA stands for intra–urban agriculture and PUA stands for peri–urban agriculture. a: Clune et al., 2017; b: Perrin et. al, 2014; c: Poore et al., 2018.

Our second method of comparing UA impacts to conventional agriculture was using generalized results from food and agriculture LCA reviews (Figure 2.10). Although these comparisons may be less precise, with different climate and local contexts, the large sample size made them more representative. The outcomes 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 0.3-0.6 kg CO_2 eq. kg^{-1} crop. Among our 157 IUA results, most of the climate change impacts were between 0.3-4.0 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 2.10a). 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 0.03-1.3 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 2.10c). 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 2.10d). 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 0.2-2 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 (Adewale et al., 2016; Foteinis and Chatzisymeon, 2016; Meier et al., 2015; Pereira et al., 2021; Poore and Nemecek, 2018). 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.

2.3.4 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 (Cleveland et al., 2017; Dorr et al., 2017; Goldstein et al., 2016b; Kulak et al., 2013; Martinez et al., 2018; Pennisi et al., 2019; Sanyé-Mengual et al., 2015a, 2015b). 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 (Figure 2.11a 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 m² ($r = 0.41$, $p = 5.4e\text{-}09$) (Figure 2.11b 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. 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

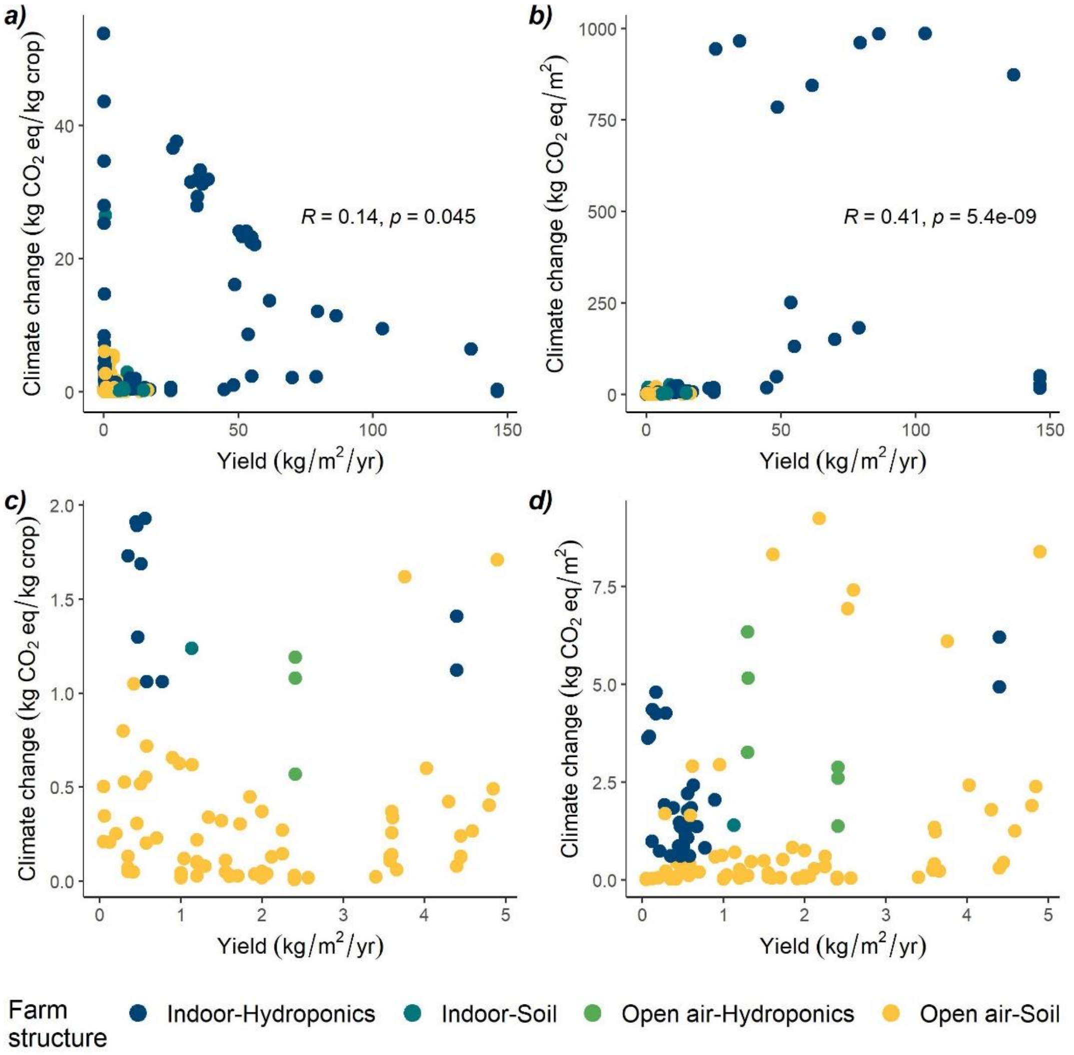

Figure 2.11 Yield was often cited by authors as a driving factor of climate change impacts within studies, where higher yields corresponded to lower impacts. Here, yields and climate change were compared for all studies (intra–urban and peri–urban agriculture) based on mass (a) and area (b), and there 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.

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 2.12). This was available for 90% of the systems studied, where it was identified by the authors, or could be interpreted using the data

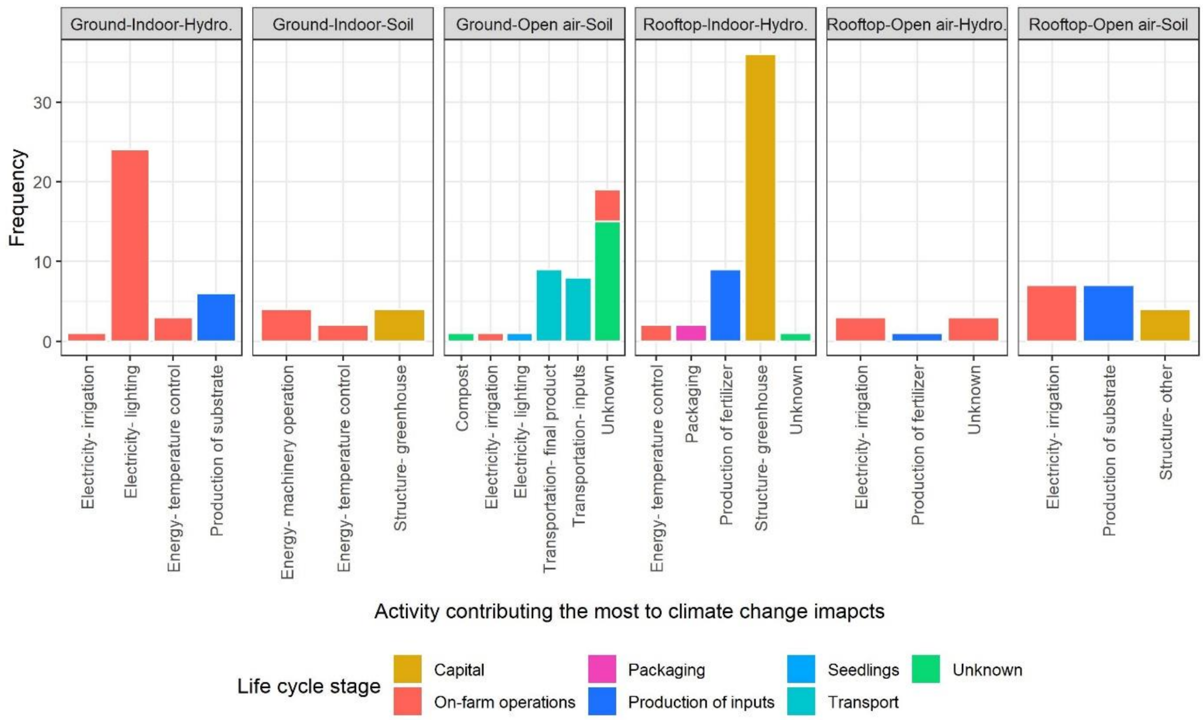

Figure 2.12 The part of the life–cycle contributing the most to climate change impacts for each system is summarized here. These are broken down into life cycle stages, which are more general and shown through the fill colors, and specific activities, which are detailed on the x-axis.

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. 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 (Boneta et al., 2019; Sanjuan-Delmás et al., 2018). 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 rooftop greenhouse hydroponics systems in Barcelona (F Corcelli et al., 2019; Llorach-Massana et al., 2017b).

For PUA (which is not shown in Figure 2.12), 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, which 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 (Clune et al., 2017; Perrin et al., 2014). 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 (Dorr et al., 2021b; Goldstein et al., 2016b; Kikuchi et al., 2018; Martin et al., 2019; Martin and Molin, 2019; Romeo et al., 2018; Weidner and Yang, 2020).

2.3.5 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 at 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 integrated rooftop greenhouse 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 2.1) 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 (Dorr et al., 2017; Kim, 2017; Pennisi et al., 2019; Sanyé-Mengual et al., 2015b, 2015a; Zhen et al., 2020), and a review on LCCs 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 (Hu et al., 2019; Liang et al., 2019; Pérez-Neira and Grollmus-Venegas, 2018; Sanyé-Mengual et al., 2015a), 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., 2020a). 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 (2020a) 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.

2.3.6 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, Rufi-Salis et. al (2020a) with 25 production systems from an indoor hydroponics rooftop greenhouse, 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 integrated rooftop greenhouse 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 (Arcas-Pilz et al., 2021; F Corcelli et al., 2019; Llorach-Massana et al., 2017b; Muñoz-Liesa et al., 2021; Rufí-Salís et al., 2020a, 2020b; Sanjuan-Delmás et al., 2018; Sanyé-Mengual et al., 2013, 2015a, 2018b; Toboso-Chavero et al., 2018).

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 only assume that there may be bias due to convenience sampling.

2.4 Discussion

2.4.1 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. 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. 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 have lower climate change impacts and low yields. Many different aspects emerged as having large climate change impacts in these systems, from

Table 2.2 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, large sources of impact. 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, appears 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, appears 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.

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.

2.4.2 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 2.1. 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 (Appolloni et al., 2021; Buehler and Junge, 2016; Guitart et al., 2012; Orsini et al., 2020; Pourias et al., 2016). 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.

2.5 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 set–up, 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.

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Chapter Three: Methodological framework for life cycle assessment of urban agriculture

3 Chapter Three: Methodological framework for life cycle assessment of urban agriculture

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A Framework for Applying Life Cycle Assessment to Urban Agriculture

Abstract

There is an increasing interest in evaluating the environmental impacts and benefits of urban agriculture (UA), especially using the method of life cycle assessment (LCA). The evidence from LCAs of UA currently available is varied, using inconsistent methods and decisions, resulting in important knowledge gaps. Here, we present a methodological framework aimed at bringing consistency and comprehensiveness to LCAs of UA. The creation of this framework was informed by a literature review, and our experience performing LCAs of diverse types of UA. The framework first puts this area of research into perspective by clarifying and proposing questions that can be addressed with LCA, and related goals. We then provide practical recommendations for performing LCAs, considering several unique aspects of UA and the resulting challenges. These include crop diversity, system multifunctionality, data availability, on-farm compost system modeling, off-farm compost system modeling, carbon sequestration, compost emission factors, creation of substrate, transport and delivery, and variability. Next, we outline directions for future research, which can improve LCAs of UA and challenge LCA in general to prompt methodological developments. This section covers alignment with other urban green infrastructure, evaluations at the city scale, combining LCA and the ecosystem service framework, and including social dimensions of UA in an assessment. Throughout the framework, we address fundamental LCA issues regarding functional units, system boundaries, uncertainty, allocation, system expansion, and social dimensions. By following this framework, future LCAs of UA can be more consistent, comparable, and holistic, and will help build knowledge and inform policy making around UA.

3.1 Introduction

Urban agriculture (UA) is a multifunctional activity with many assumed and demonstrated benefits for cities and their inhabitants. These social, economic, and environmental benefits position UA as a powerful tool to improve sustainability of cities, and, due to its food function, to possibly improve sustainability of urban food systems (Azunre et al., 2019; Gómez-Villarino et al., 2021). It may help minimize environmental damage from the highly impactful conventional food system, from site-specific pollution to diffuse greenhouse gas emissions (Nicholls et al., 2020; O'Sullivan et al., 2019). While the former can be measured using in-situ measurements, environmental impact models are used for the latter, such as life cycle assessment (LCA). Such assessments are complex and demanding, and although the method is standardized and provides reliable results, the available LCAs of UA have highly variable findings (Dorr et al., 2021a). Currently, there is no consensus on important questions around the environmental performance of UA, such as: what types of UA have lower impacts than others; what are the main sources of impacts in UA; and can UA help reduce environmental impacts of the food system? This lack of consensus can be largely attributed to the inconsistent use of LCA for UA, in terms of system modeling decisions, system boundaries, and reporting.

Methodological frameworks are structured, practical guidelines that improve consistency and quality of methodological approaches (McMeekin et al 2020). These are the operational outputs of reflection, advancement, and maturation of a method. The general procedure for developing a methodological framework is to identify evidence to inform the framework, develop it, and test/refine it (McMeekin et al 2020). LCA frameworks are based on critical reviews of LCA case studies of a given sector, and build upon previous methodological frameworks. They describe how LCAs of that topic are usually done, and recommend how they should be done in the future.

General LCA frameworks have been proposed to improve the rigor and cohesion of LCAs, and include the International Standardization Organization (ISO) framework (ISO, 2006a, 2006b), the ILCD handbook (European Commission, 2010a), and the Product Environmental Footprint Category Rules Guidance (European Commission, 2017). Frameworks for LCA of specific sectors highlight unique aspects that should be considered in a particular way. Such frameworks have been done for diverse topics, from waste management (Laurent et al., 2014b) to bioplastics (Bishop et al., 2021). Agriculture has been the focus of much LCA framework and methodological research. This research includes carbon footprint guidelines for crop production (Adewale et al., 2018), suggestions for improving LCAs of organic agriculture (van der Werf et al., 2020), recommendations for fruit orchard LCAs (Cerutti et al., 2014) and vegetable LCAs (Perrin et al., 2014), and frameworks for evaluating agricultural practices with LCA like climate smart agriculture (Acosta-Alba et al., 2019) and microbial inoculants (Kløverpris et al., 2020). Other work has evaluated the combination of agricultural LCAs with circular economy (Stillitano et al., 2021) or ecosystem service assessments (Tang et al., 2018). Not to mention the large body of work reviewing the methodological choices, challenges, and best practices of agricultural LCAs in general (Audsley et al., 1997; Brentrup et al., 2004; Caffrey and Veal, 2013; Cucurachi et al., 2019; Dijkman et al., 2018; Mclaren, 2010; Nemecek and Gaillard, 2010; Notarnicola et al., 2017, 2012). Such methodological reflections and frameworks have not yet been done for LCAs of UA.

This framework begins by reflecting on the goals and expectations of doing LCAs of UA, followed by practical recommendations for performing an UA LCA, and research directions for improving UA LCAs. It presents the challenges of including certain aspects of UA in LCA, reviews how these aspects are treated in LCAs of UA or other relevant topics, and recommendations for how to treat them going forward. This framework is based on our literature review and meta-analysis of UA LCAs (Chapter 2) (Dorr et al., 2021a), expert opinion, and examples of application of LCA to similar topics. It is intended to build upon frameworks of agricultural LCAs, and many general issues of agriculture LCAs are relevant here, but were not presented if they are not specific to/exemplified by UA. We cover a broad range of UA here, and consider UA defined as “food production in and around cities”.

3.2 Why do life cycle assessments of urban agriculture?

The goal of an LCA will define how the assessment is set up. All future decisions regarding system boundaries, functional unit (FU), and interpretations should be consistent with the defined goal of the study. The goal should reflect the pursuit of a larger question. Larger questions around UA that may be evaluated using LCA are described in Table 3.1.

Table 3.1 The goal of a life cycle assessment should respond to a larger question. Some relevant questions for life cycle assessment of urban agriculture are presented here, along with the suggested functional unit and description/justification for each question. Some questions are already prevalent in the literature, and some are our original suggestions and have not been addressed before.

Question	FU	Description
Is UA an environmentally positive type of green infrastructure to implement in a city?	Area	Green infrastructure is promoted in cities, and many types are possible. City leaders must decide which types to implement.
Is UA an environmentally positive way to feed the city?	Product	In light of new urban food planning strategies, and initiatives to reduce impacts of public food procurement, we should investigate if UA is a useful strategy.
Is UA a meaningful way to reduce a city's GHG emissions?	Area, product, consumer	Cities have pledged to reduce greenhouse gas emissions, which UA may address through land use, replacing other food sources, changing consumers' behaviors, or altering organic waste treatment.
If we do UA for social/non-environmental reasons, is it at least not very environmentally harmful?	Product, area	Many UA projects do not claim to have environmental motivations or particularly low impacts, but they are promoted based on other merits. Are there important tradeoffs between the social and environmental dimensions? Can we justify an environmentally harmful activity if it delivers social benefits?
Which type of UA should be developed/promoted for a given objective (indoor or outdoor, hydroponics or soil-based, commercial or non-profit, professional or volunteer-based…)?	Area, product	Developers, city leaders, and stakeholders may have land that they want to dedicate to UA. With the vast diversity of types of UA, they may need support deciding which type to develop, according to environmental and other dimensions.
How can UA be designed or managed to minimize environmental impacts?	Area, product	In many cases, UA will be practiced regardless of the above questions. Then, we should inform practitioners of the best practices to minimize their impact.

A common goal of UA LCAs is evaluating the environmental impacts of urban food production at the farm-scale, which involves a rather straightforward attributional LCA. In this case, the impacts of all processes at the farm or garden are evaluated. This goal is appropriate for determining benchmark values, which can be used for other assessments such as scaling up UA in a city. Transparency in system modeling is particularly essential here, because the outcomes of these studies are likely to be used by others. It is also useful to highlight if the system is an innovative or research system, so that users know that it is not representative of common types of UA and/or is not optimized, and will not make extrapolations or general assumptions based on those results. This is a very common goal in UA LCAs, but is often not the only goal, and is combined with one of the following goals.

LCA can be applied to UA to identify ways to improve the environmental performance of a farm or garden (Dorr et al., 2021b; Martin et al., 2019). This usually begins with the above-mentioned straightforward assessment of food production at the farm-scale, and evaluates some alternative, usually hypothetical, scenarios and practices that may be implemented in the system. This is useful to indicate the best path forward for systems looking to reduce environmental impacts, or to evaluate whether potentially complicated interventions are worthwhile from an environmental perspective.

The goal of an UA LCA can be to compare urban and rural agriculture (Fisher and Karunanithi, 2014b; Kulak et al., 2013). This can be done with a land- or product-based FU, which are the two most common FUs in rural agriculture LCAs since they represent two main functions of agriculture: growing food, and land management (Notarnicola et al., 2017). In this comparison, it is strongly not recommended (and irrelevant) to use inventory data from rural agriculture for the UA systems. It is especially important to include the steps of delivery to the market or consumer, since proximity to the consumers is a main distinction of UA.

Another goal might be to compare an UA system with an alternative urban land use or green infrastructure (F. Corcelli et al., 2019; Kim et al., 2018). Here, a land-use perspective is taken rather than a product-based perspective, and therefore an area-based FU is more appropriate than a product-based FU. UA could be compared to, for example, a green roof, rooftop solar panels, lawns, or golf courses. System expansion should be done to include the multiple functions of systems, when relevant and quantifiable: for UA this is the production of food, and for solar panels this is the production of energy. For non-productive systems like parks and green roofs, the additional functions (recreational, pleasure) are difficult to quantify and will likely not be accounted for. In this comparison, it may be acceptable to use inventory information from rural agriculture, although it is preferable and more precise to have data from UA. So far, relatively few UA LCAs have adopted this goal.

A goal may be to compare different types of UA (Goldstein et al., 2016b; Ledesma et al., 2020b; Sanyé-Mengual et al., 2015b). Either land or product based FUs would be appropriate here (as is done with rural agriculture LCAs). Using both FUs is preferable, since it has been well-documented that the relative performance of rural farms can depend on whether a product- or area-based FU is used (as in the case of organic vs conventional agriculture) (Caffrey and Veal, 2013; Meier et al., 2015). Such an assessment should generally follow a similar model as the assessment of food production at the farm-scale. If the compared forms of UA provide different, quantifiable functions in addition to food production—such as building insulation or reduced stormwater runoff—the multiple functions can be accounted for using system expansion and avoided burdens. If the compared systems have some identical processes, such as delivery schemes, they can be omitted in the comparison. However, it is useful to include them if possible, to allow for a more complete LCA which can be used by others, since UA LCAs are scarce.

Finally, a more complex goal may be to evaluate the effects of UA on specific external outcomes in a consequential LCA model. This may include the effects on urban transport logistics, consumption habits of UA participants, urban organic waste management, or lawn maintenance (Cleveland et al., 2017; Oliveira et al., 2021; Puigdueta et al., 2021; Weidner and Yang, 2020). This type of goal allows for a great variety of system modeling choices. These may include the impacts of operating UA plus the consequences (i.e., on transport logistics…), or include only the consequences, but transparency in this decision is essential. If impacts of operating UA are not included, then it is inappropriate to make general conclusions about the net impacts of UA. This type of assessment is particularly situational, and it is difficult to extrapolate results of such a study to other contexts. This strategy is especially interesting to isolate one aspect of UA and compare it to business as usual or other scenarios, but researchers must take caution in considering a possible cascade of effects.

3.3 Practical recommendations for UA LCAs

In this section, we describe unique aspects of UA that present methodological challenges for LCAs, and our recommendations for addressing them. Each section includes an explanation of the challenge, examples of how it has been treated in previous urban or rural agriculture LCAs, and our recommendations for treating it going forward. Section 3.4, on compost, has a different layout with subsections because there are numerous challenges, and to the best of our knowledge its inclusion in agricultural LCAs has not been reviewed before.

3.3.1 Crop diversity

Challenge:

The most common FU in crop production LCAs is mass of crop (Notarnicola et al., 2017). For agricultural systems that cultivate one crop, there are no allocation issues: all inputs and impacts can be assigned to the one crop. For farms that grow multiple crops either with temporal diversity (crop rotation) or spatial diversity (polyculture/intercropping), inputs or impacts must be allocated to a single crop (Adewale et al., 2018). For polycultures, rural/professional farmers can often specify which inputs were used in the various parcels on the farm, and fixed inputs can be allocated by mass, revenue or other measure (Caffrey and Veal, 2013). For crop rotations, various allocation principals have been proposed (Brankatschk and Finkbeiner, 2015). It is difficult to allocate inputs to one crop in UA, where crop diversity is often very high: studies suggest that urban farms and gardens may cultivate on average 20-30 crops per year, with extremes of 80-130 (Gregory et al., 2016; Kirkpatrick and Davison, 2018; Pourias et al., 2016) (Chapters 1 and 4.1). It is unreasonable to expect urban farmers and gardeners to distinguish their inputs between such a large number of crops, so LCA practitioners may need to account for production of many different products in one FU. This issue is not unique to UA—it is also relevant for diversified rural farms and community supported agriculture (CSA) (Caffrey and Veal, 2013; Christensen et al., 2018)—but it is likely especially extreme in UA.

Examples:

In many UA LCAs dealing with a large number of crops, FUs were chosen that were not based on the mass of product, such as annual production or land area (Martinez et al., 2018; Pérez-Neira and Grollmus-Venegas, 2018; Sanyé-Mengual et al., 2018a). This is a useful way to avoid allocations with high uncertainty, consider additional functions of agriculture, and to compare systems within a study. A downside is that results are difficult to extrapolate, since diverse and sometimes unknown crops are produced. Another strategy was to use data from the literature or estimates from farmers to construct an expected LCI for each crop (Caputo et al., 2020; Kulak et al., 2013; Liang et al., 2019; Weidner and Yang, 2020). This provides useful results for each crop in complex systems, but accuracy is lost when using estimates or data from other potentially different systems. Furthermore, when this data comes from rural agriculture, the resulting LCA probably does not reflect many specificities of UA. Other researchers have allocated between many crops based on mass, area, calorie content, or nutritional index, or time of cultivation of each crop (Pennisi et al., 2019; Rufí-Salís et al., 2020a; Sanyé-Mengual et al., 2015b), which generates results per crop that can be used elsewhere. Finally, some researchers did not differentiate between the multiple crops, and used a FU of mass of mixed crops (Boneta et al., 2019; Hu et al., 2019)(Chapter 4.1). These results are difficult to use elsewhere, since unique mixes of crops are not precisely comparable. LCAs of rural farms with many crops, such as community supported agriculture (CSA), have also used FUs of kilogram of mixed crop (Christensen et al., 2018; Pepin, 2022).

Recommendations:
In summary, the main options for dealing with multi-crop UA systems are to evaluate a basket of products (by mass or by converting to calories or nutritional indexes), allocate between products, or choose a FU that is not based on production. There is no clear best strategy, and ultimately the choice of FU depends on the goal of the LCA. When a FU other than single crop is used, a breakdown of how much of each crop was grown should be provided, to give some indication of what the food outputs of the system were.

3.3.2 Multifunctionality

Challenge:
Nemecek and Gaillard (2010) outlined three main functions of agriculture that can be evaluated with three different FUs: food production, with a product-based FU; land management with a land-based FU; and commercial objectives, with a revenue-based FU. In UA, we can add a critical fourth category around social functions, including well-being, education, beautification, recreation, and more. In reality, many UA projects have important objectives across multiple functions, and it is precisely this multifunctionality that is argued to be one of the main assets of UA (Artmann and Sartison, 2018; Gómez-Villarino et al., 2021; Wadumestrige Dona et al., 2021). This multifunctionality poses a problem for choosing a FU that represents the function of the system, and for accounting for co-products and co-services.

This also highlights a problem in reporting, where the multiple functions of an urban farm or garden are not usually well described in LCAs, preventing readers from qualitatively understanding the functions of systems. Indeed, the LCI and system descriptions usually provide a clear picture of the technical aspects of an UA case study, but other characteristics—such as the objective of the site, commercial status, who does the farming/gardening, whether land and water are paid for—are often overlooked (Chapter 2).

Examples:

A main strategy for dealing with multifunctional systems is calculating and interpreting results based on multiple FUs (European Commission, 2010a). Indeed, relative performance of agricultural systems can change dramatically when using a mass-based or land-based FU, as seen in Chapter 4.1 and elsewhere (Haas et al., 2000; van der Werf et al., 2020). Use of multiple FUs has been somewhat implemented in UA LCA, although most studies (about 70%) use only one FU (Chapter 2) (Dorr et al., 2021a). In 50% of UA LCAs, mass of crop is the only FU used. Other FUs in UA LCAs include land, annual operations, annual food consumption, calories, and revenue (Chapter 2). No LCA has used a FU that reflects the social function of UA.

A unique strategy to account for multiple functions of rural agriculture by Boone et al. (2019) combined an ecosystem service assessment and LCA. They allocated impacts between the various ecosystem services provided by rural farms, effectively isolating the food production function. This could be done for cultural ecosystem services as well. System expansion and substitution are used in agricultural LCAs when the multiple purposes/outputs are quantifiable, but this is less relevant for UA with many intangible functions (Caffrey and Veal, 2013).

Recommendations:
A better consideration of the multiple functions of UA in LCA is necessary, first through the use of multiple FUs. Plus, case studies should be characterized more holistically, with more information about the objectives and additional outcomes of UA case studies, rather than simply technical descriptions. In a more concrete step, ‘alternative’ LCA methods should be explored to quantify the social functions of UA (detailed more in section 3.4.4). Some relevant FUs to develop here include number of people from the public engaged in a year, number of volunteer hours, or person-hours of education.

3.3.3 Data availability at urban farms and gardens

Challenge:
Data collection is widely considered the most labor-intensive step of any LCA, due to the high data demands of the method. For an agricultural LCA, data are needed regarding the inputs and outputs of food production at the farm. In more traditional, professional, and commercial agriculture, such primary data can come from interviews with farmers, purchase or sales records, or making estimates/calculations (Christensen et al., 2018). Secondary data can be used to fill in missing information, or create entire inventories, thanks to agricultural censuses and research, such as the UC Davis Cost and Return Studies (Caffrey and Veal, 2013). For UA, such data are not readily available or estimable because urban farmers and gardeners usually do not keep records (Cleveland, 1997; Egerer et al., 2018; Whittinghill and Sarr, 2021). Evidence shows that inputs and food production in UA can be extremely variable and difficult to predict, casting doubt on the relevance of using secondary data for UA production (Chapter 1). Even when farmers and gardeners are willing to collect this data, it will likely come in various units and methods, which researchers must convert and consolidate (Christensen et al., 2018). Collective and community-based UA may have many participants who harvest and use inputs, resulting in decentralized and challenging data recording. Self-reporting and participatory methods face issues of reliable and consistent data collection (CoDyre et al., 2015).

Examples:

The available UA LCAs are based on both primary and secondary data. Data for UA LCAs come from many different sources, including directly measured data, operations records, farmer and gardener interviews and surveys, and secondary data from urban or rural agriculture (Chapter 2) (Dorr et al., 2021a). Data sources and data collection difficulties are largely discussed in research on UA practices in general, but not so much in UA LCAs (McDougall et al., 2019; Pollard et al., 2018a).

Recommendations:
Due to the variability and lack of data regarding UA practices, collecting primary data from case studies should be prioritized. Data from past records of operation may be used, although it is unlikely that urban farmers/gardeners have records of all necessary information for an LCA. A data collection campaign, with commitment from farmers/gardeners, may be necessary. Researchers are encouraged to discuss the data needed with farmers and gardeners early on, identify the most feasible methods to collect it, create a data collection plan, and regularly follow up to ensure reliability. This is a crucial step because if it is unclear, or is unreasonably burdensome on farmers/gardeners, then data collection may be abandoned or incomplete. Researchers should consider the types of data that may already be collected at urban farms and gardens (i.e. level of detail, units), and adapt the data collection plan to make use of it. Surveys, growing logs, and harvest notebooks should be designed with farmers/gardeners to track harvest and inputs (Nicholls et al., 2020)(Chapter 1). Water use should be measured using water meters (several studies mention the Gardena® water meter (Egerer et al., 2018) (Chapter 4.1), or calculated using the number of buckets or watering cans used and their volume (Pollard et al., 2018a). Attention should be paid to ensure methods account for leaks in irrigation systems, which may be substantial (Chapter 1). Inputs like compost and fertilizers should be tracked through the amount applied, or the amount purchased/delivered (although this may need to be temporally allocated to fit the time frame of the study). The detailed description of our data collection methods with UA case studies in the appendix of Chapter 4.1 provides many concrete examples of how to collect data with diverse systems.

3.3.4 Compost
Compost is a main input to many urban farms and gardens (Cofie et al., 2006; Dobson et al., 2021; Edmondson et al., 2014) (Chapters 1 and 4.1). A proposed environmental advantage of UA is its potential to take up urban waste and reuse its organic matter and nutrients in the form of compost to grow food, rather than incinerate or landfill it (Goldstein et al., 2016a; Mohareb et al., 2017; Specht et al., 2014; Weidner and Yang, 2020). As a result, compost is central to UA discourse, although its use and impacts have been infrequently and inconsistently quantified for UA LCAs (Dorr et al., 2021a). Even for rural agriculture LCAs, compost is often omitted, or its inclusion is inconsistent and unclear (Bartzas et al., 2015). Surprisingly, compost is not explicitly mentioned in LCA reviews of organic agriculture, where it is expected to be more important, as opposed to conventional agriculture (Meier et al., 2015; van der Werf et al., 2020). LCAs that have focused on compost use in agriculture find that the greenhouse gasses emitted from microbial decomposition are a major contributor to climate change impacts, and that system modeling decisions around avoided burdens and allocation have large effects on the results for rural agriculture (Bartzas et al., 2015; Christensen et al., 2018; Martínez-Blanco et al., 2009) and for UA (Dorr et al., 2017; Liang et al., 2019; Martin et al., 2019) (Chapter 4.1). Therefore, compost is given extra attention for this section.

Compost is complicated to evaluate using LCA because there are multiple perspectives that can be taken—waste treatment and agricultural input production—and because its production is a multifunctional process, therefore requiring important system modeling decisions (Martínez-Blanco et al., 2011). Also, there is large variability in emission factors for composting, which usually account for the largest share of impacts. Here, we first briefly characterize compost use in UA, ascertaining its importance and justifying its further consideration for UA LCA. Then we present the challenges, examples, and recommendations for system modeling decisions; and commonly used yet variable emission factors for composting inventories.

3.3.4.1 *Characterization of compost use in UA*

Compost is the most common input in many forms of UA. Surveys of UA practices have found that 80-95% of urban gardens and allotment plots made and/or used compost (Dobson et al., 2021; Edmondson et al., 2014; Guitart et al., 2015; Wielemaker et al., 2019). For rooftop farms, this was 62% (Appolloni et al., 2021), and on the lower end, the estimate is 30-35% (Dewaelheyns et al., 2013). In contrast, about 35% of the 16,585 rural organic farms in the USA used compost in 2019 (most used manure instead) (USDA, 2020).

In terms of quantity of compost applied, it seems that UA can use much larger amounts than rural agriculture. Estimates of typical annual compost application rates in rural agriculture range from 2-45 tonnes/ha (BioCycle, 2004; Erhart and Hartl, 2010; Rittenhouse, 2015; Schwarz and Bonhotal, 2016; Van der Wurff et al., 2016). Such average estimates have not been summarized for UA, but case studies report extremely large amounts of 52 tons/ha (Martínez-Blanco et al., 2009), 58 ± 70 tons/ha (Chapter 1), and 90-170 tons/ha (Chapter 4.1). More moderate use of 2-12 tons/ha has been found in other urban farms (Grard et al., 2022; Sanyé-Mengual et al., 2015b). It is difficult to generalize without a systematic evaluation, but it appears that among farms that do use compost, UA sites usually use a rate as large or larger than rural sites. This may be explained because urban soils require large inputs to become fertile, or because substrates must be created for UA (Beniston et al., 2016; Gregory et al., 2016).

It is unclear what the typical source of compost for UA is. The source of compost should have important impacts on the system modeling decisions, as detailed below. Some studies find that compost in UA is mostly made on-site at the farm/garden (Dewaelheyns et al., 2013; Dobson et al., 2021), while others suggest it is mostly purchased compost made off-site (McDougall et al., 2019).

3.3.4.2 *Off-farm compost system modeling*

Challenge:

Off-farm compost refers to the compost purchased from municipal or industrial composting facilities, as opposed to on-farm compost, described in the following section. Off-farm compost is a recycled input, similar to using recycled plastic materials or recycled paper. This is an example of ‘open loop’ recycling, because the recycled product does not re-enter the system that produced it (ISO, 2006b). Accounting for recycled inputs is a distinct type of

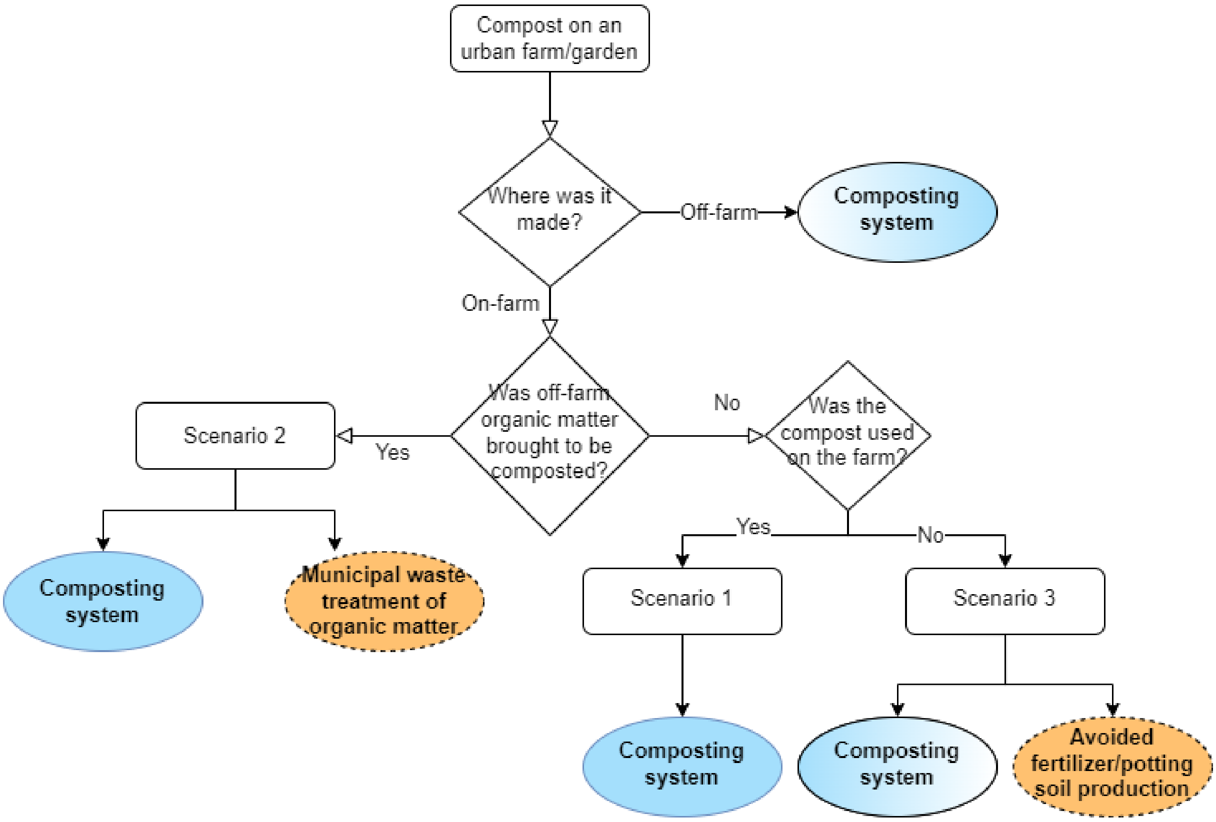

Figure 3.1 A decision tree to clarify the different scenarios of composting for an urban farm or garden, and how to account for composting impacts. Blue circles represent impacts from composting emissions, and orange circles with dotted outlines represent substituted processes that can be subtracted from the farm/garden's impacts, thanks to composting. Blue circles with gradients represent the fact that not all impacts from composting in that scenario will go to the farm/garden: they should be allocated between the organic waste producer and the compost user. The numbered scenarios are detailed in section 3.3.4.3.

allocation issue, and is a longstanding, complicated, and contested topic in LCA (Frischknecht, 2010; Huppes and Curran, 2012; Toniolo et al., 2017; Weidema, 2000).

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

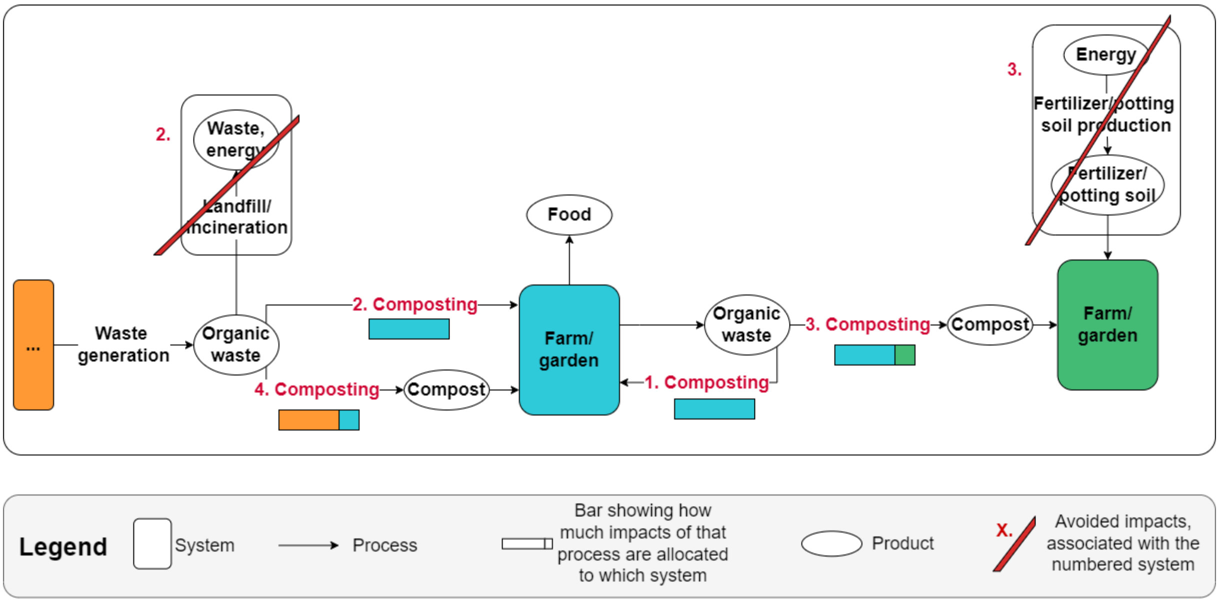

Figure 3.2 A process diagram shows the different composting scenarios as described in the text, from the perspective of the blue farm/garden in the center. The numbers refer to the scenarios described in section 3.3.4.3, and scenario 4 refers to off-farm compost, described in section 3.3.4.2.

Commission, 2010b; Guinée et al., 2004). For compost, this has been done using the relative revenue at a composting plant between waste dumping fees and compost purchases (Christensen et al., 2018; Pepin, 2022).

For UA LCAs where off-farm compost was used, such system modeling decisions have been mixed. In most cases, off-farm compost was included using the simple cut-off approach, giving all impacts to the compost product, with no avoided burdens (Goldstein et al., 2016b; Ledesma et al., 2020b; Liang et al., 2019; Martin et al., 2019; Rothwell et al., 2016). Dorr et al. (2017) credited systems using compost with avoided waste treatment because they compared the impacts of compost-based and peat-based substrates.

Recommendations:
We recommend treating off-farm compost as a recycled input, using the refined cut off method to give compost no impacts from the waste material production, and some impacts from the composting process (Figure 3.1 and 3.2). Impacts from composting should be allocated between compost production and organic waste treatment.

3.3.4.3 *On-farm compost system modeling*

Challenge:
On-farm compost refers to the composting operations in a farm/garden, dedicated mostly to composting inedible plant biomass from cultivation. There are several possible scenarios for on-farm compost and consequently several modeling options (Figure 3.1 and 3.2). On-farm compost may be:

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

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

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

These possible scenarios, and the relevant system modeling for LCA, have not been explicitly examined before. Thus, there is a lack of clarity in the literature regarding such scenarios and decisions.

Examples and recommendations:
Scenario 1 is a type of 'closed loop' recycling system, where the waste is generated and the recycled product is used within the same system. Examples of this are in Boneta et al. (2019) and Sanyé-Mengual et al. (2015b). System modeling is rather straightforward here, as no allocation is needed (ISO, 2006b). In this situation, all impacts from composting should be given to the farm/garden, and no avoided burdens or allocation should be done.

In scenario 2, composting is no longer a closed-loop system, because waste enters the system from elsewhere and is treated at the farm/garden. Here, the farm/garden serves two functions: growing food, and treating waste. The additional function of avoided municipal waste treatment of biomass brought to the farm should be accounted for. Allocation is likely not possible here, because amounts of off-farm and on-farm organic waste cannot be accounted for. Then, the additional waste-treatment function should be accounted for through system expansion and substitution, by subtracting impacts of the alternate fate of organic waste from the UA system. This results in environmental credits to the UA system. This type of scenario is demonstrated in our case studies (Chapter 4.1).

Scenario 3 composting can be found at urban farms that create inedible biomass waste (all farms) but do not use compost, such as soilless hydroponics or aeroponics systems. This type of composting represents a multifunctional process: it treats the farm's waste and creates a recycled material to be used elsewhere. Here the UA site can be viewed as the waste generator, as discussed in the off-farm compost section (section 3.4.2). Farms and gardens should be credited with avoided environmental burdens from production of the fertilizer or potting soil that the produced compost can substitute (F. Corcelli et al., 2019; Goldstein et al., 2016b). A review and guidelines for including this in an LCA was done by Vieira and Matheus (2019). Composting for waste treatment of biomass can account for 10-15% of climate change impacts (F. Corcelli et al., 2019; Sanjuan-Delmás et al., 2018), but avoided burdens of fertilizer production can result in this process having a net positive impact (F. Corcelli et al., 2019).

3.3.4.4 *Carbon sequestration*

Challenge:
Compost is rich in organic carbon, and upon application to soil this carbon can be stabilized and stored in the soil (Lal et al., 2015). Carbon sequestration from compost is seen as a promising way to remove greenhouse gasses from the atmosphere and tackle climate change (Tiefenbacher et al., 2021). From an LCA perspective, this can be seen as avoided climate change impacts, where farms using compost should receive environmental credits for the equivalent CO_2 sequestered as organic carbon in their soils. The challenge here is that the ecological processes of carbon sequestration in soils are complex and poorly understood. This results in high uncertainty in a process that can largely influence the LCA results (Mclaren, 2010; Strohbach et al., 2012; Tidåker et al., 2017). Soil carbon models are available, but are usually highly time and data intensive, and are poorly adapted to UA with its potentially unique substrate and high compost amendments (Dorr et al., 2017). Another complicating factor is the time scale of carbon sequestration and LCAs. Most LCAs model climate change

impacts of a system over 100 years, and provide the average impact over the given time scale (for agriculture, often this is 1 year or one growing season). Experimental measures of soil carbon sequestration are only available on a 10-20-year time scale, and many measures are less than 5 years. The ultimate, long-term fate of organic carbon is mostly unknown and highly context dependent.

Examples:
Inclusion of soil carbon sequestration from compost in LCAs has been mixed, and in UA LCA specifically has been limited. Several researchers argue for including it in agricultural LCAs (Adewale et al., 2018; Martínez-Blanco et al., 2013), and others claim it is too poorly understood to be meaningfully considered and should be excluded (Joint Research Centre, Institute for Environment and Sustainability, 2012; Nordahl et al., 2022). Some compost LCAs (from a biowaste treatment perspective) have included carbon sequestration at rates of 10-14% of organic carbon (Boldrin et al., 2010; Tonini et al., 2020; Vaneeckhaute et al., 2018). Dorr et al. (2017) used a soil model, applied to UA compost-based substrate and potting soil, and estimated that carbon sequestration benefits were small, offsetting 0.2-3% of GHG emissions of the farm. In our case studies (Chapter 4), most of which used particularly large amounts of compost, compost carbon sequestration offset 3-23% of climate change impacts. Many agriculture LCAs explicitly omit carbon sequestration from compost because of high uncertainty (Christensen et al., 2018; Seufert and Ramankutty, 2017), while some include it (Rothwell et al., 2016), and some present impacts with and without carbon sequestration (Dorr et al., 2021b)(Chapter 4.2). LCAs of other urban green infrastructure, such as parks and golf courses, usually include carbon sequestration, and it often largely affects the results, even resulting in the entire system acting as a carbon sink (Bartlett and James, 2011; Nicese et al., 2021; Strohbach et al., 2012).

Recommendations:
We recommend not including carbon sequestration from compost (or other organic inputs) in the main results of UA LCAs, due to the large uncertainties. It can be included in sensitivity analyses, or secondary results, to explore the extent to which it may be important, but care must be taken to highlight the uncertainty in those results.

3.3.4.5 *Compost emission factors*

Challenge:
The most impactful component of the compost life cycle is gaseous emissions of methane, nitrous oxide, ammonia, and volatile organic compounds during the composting process (Boldrin et al., 2009; Pergola et al., 2020). The impact categories affected most by gaseous emissions are climate change, acidification, eutrophication, and photochemical ozone formation (Pergola et al., 2020). High variability in gaseous emissions from composting—due to differences in technical systems, input material, and composting practices—result in high variability in impacts from compost (Joint Research Centre, Institute for Environment and Sustainability, 2012). As a result, it is very difficult to summarize generic emission factors. Ideally case-specific inventory data should be used, but that is not often available (Boldrin et al., 2009). Such variability in inventory data for a main input of UA poses a problem for choosing composting processes to use in LCAs.

Examples:

Many UA LCAs use composting emission factors from Andersen et al. (2012, 2011), Martínez-Blanco et al. (2010), and Colón et al. (2010), because they measured inventory data specifically for home composting, which can be expected to be more similar to small scale, on-farm composting operations. The LCA database Ecoinvent (Wernet et al., 2016) is also a main source of composting LCI data in agricultural LCAs. The process in the database comes from Edelmann and Schleiss (1999). Table 3.2 shows the wide range in composting greenhouse gas emission factors from these major data sources for agricultural LCAs, among others. Although it is far from an exhaustive list of emission values, it highlights the potential pitfalls from selecting composting inventories with such variability. Indeed, in our case studies we found that climate change impacts for the farms were reduced by 2-14% when we used the inventory from Ecoinvent rather than from the review by Nordahl et al. (2022). For more complete summaries of measured composting emission factors, see reviews papers by Nordahl (2022), Boldrin (2010), and Amlinger (2008), and discussion section reviews in Quiros (2015) and Avadi (2020).

Recommendations:
To address the variability of composting emission factors for UA LCAs, we recommend modeling multiple scenarios with different emission factors when large amounts of compost are used in a system. Emission factors can be chosen from a specific source with a representative composting technology, as shown in Table 3.2, or averages of multiple sources can be used. Monte Carlo simulations can be performed to include a distribution of composting emission factors and obtain a range of results.

3.3.5 Creation of substrate

Challenge:
A unique characteristic of UA, as opposed to rural agriculture, is that it is not necessarily carried out on soil. Soil, or top-soil, is defined as natural bodies made of organic and inorganic material that are formed at the surface as the result of complex biogeochemical and physical processes (Brevik and Arnold, 2015; Hartemink, 2016). Although many UA sites grow directly in the soil, this is often not an option due to soil pollution in urban areas, or lack of greenfields. In these cases, soilless cultivation methods are used (such as hydroponics, aeroponics, or aquaponics), or a substrate/growing medium may be created. Substrate is a unique input that has not been considered in agricultural LCAs. This represents a kind of infrastructure, which requires large volumes of material inputs, with a large variability of possible materials. Current practices around creation of soil/substrate in UA LCAs are unclear, because authors often do not describe the nature of the substrate, it seems to be inconsistently included, and system modeling decisions around recycled materials are variable (Dorr et al., 2021a). This creates a challenge for clarity and consistency in UA LCAs.

As a type of fixed input and infrastructure, the lifetime of substrate will directly affect its impacts, but very little information is available regarding the expected or actual lifetimes of substrate in UA. A main determining factor in the lifetime of infrastructure is the durability of the material itself. Since substrate will likely be amended and used indefinitely, rather than becoming degraded and discarded, this is probably not the limiting factor. Rather, substrate lifetime will likely be determined by the lifetime of the UA project itself or the building it is located on (Romanovska, 2019). There are few records of the lifetime of UA projects, but given UA's sometimes transient or uncertain economic nature, there is reason to suspect that such lifetimes may be shorter than anticipated (Demailly and Darly, 2017).

Reference	Type of composting system	N_2O emissions	CH_4 emissions	GHG emissions	Notes (CO, NH_3, VOC emissions)
Andersen 2010[a]	Home composting, closed unit	0.30-0.55	0.4-4.2	100-239	6 composting units
Martínez-Blanco 2010 HC[b]	Home composting bin, mixed	0.676	0.158	205.4	VOCs = 0.559, NH_3 = 0.842.
Martínez -Blanco 2010 IC[b]	Tunnel composting, with biofilters for fugitive gas	0.092	0.034	28.3	VOCs = 1.21, NH_3 = 0.11.
Colón 2010[c]	Fruit and vegetable scraps, yard waste, home composting	0.2	0.3	67.1	VOCs = 0.32, NH_3 = 0.03.
Quirós 2014 HE[d]	Home composting, high-emission system	1.16	1.35	379.4	Leftover fruits and veg, yard waste. NH_3 = 1.3.
Quirós 2014 LE[d]	Home composting, low-emission system	0.2	0.295	67.0	Leftover fruits and veg, yard waste. NH_3 = 0.03.
Ecoinvent v3.5[e]	Open windrow composting	0.025	1	32.5	Retrived from Ecoinvent.
AgriBalyse- GW[f]	Green waste	0.48	0.21	148.3	Green waste. VOCs = 0.14, NH_3 = 1.87
AgriBalyse- BW[f]	Bio waste	0.13	1.15	67.5	Biowaste. VOCs = 0.21, NH_3 = 6.23
Nordahl 2022 YW[g]	Yard waste, average from review	0.0432	2.31	70.6	Average of 9 values
Nordahl 2022 OFMSW[g]	OFMSW, average, from review	0.068	0.879	42.2	Average of 21 and 19 values for CH_4 and N_2O
Nordahl 2022 manure[g]	Manure, average, from review	0.354	2.82	176.0	Average of 41 and 45 values for CH_4 and N_2O

Table 3.2 Emissions of N_2O, CH_4, and the sum of greenhouse gas (GHG) equivalents for N_2O and CH_4 are shown in kilograms of emission per ton of fresh waste composted, from some of the main sources of composting emission factors for urban agriculture life cycle assessments. GHG emissions are presented in kilograms of CO_2 eq. OFMSW: organic fraction of municipal solid waste. a) Andersen et al., 2010, b) Martínez-Blanco et al., 2010, c) Colón et al., 2010, d) Quirós et al., 2014, e) Wernet et al., 2016, f) Asselin-Balençon et al., 2020, g) Nordahl, not yet finished.

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Examples:
Substrate is created from many different materials. The dominant organic materials that make up growing media worldwide are peat, coir, wood and compost (Barrett et al., 2016). In UA, materials such as crushed brick, spent coffee grounds, spent brewer's grain, and shredded paper have been used (Dorr et al., 2021b; Grard et al., 2020; Martin et al., 2019). Several LCAs of rooftop UA and green roofs have found that creating and replenishing substrate was the largest contributor for most impact categories (Dorr et al., 2017; Kim et al., 2018; Vacek et al., 2017).

There are not many LCAs about substrate materials (Quantis, 2012; Toboso-Chavero et al., 2021), and even fewer that focus on the substrate choices integrated at the farm production scale (Barrett et al., 2016). Evaluation at the farm scale (as opposed to at the growing media production scale) is important because the type of substrate will likely affect other parameters, such as yield, water use, and fertilizer use (Barrett et al., 2016).

The numerous possible types of substrate lead to many options for modeling the materials. The LCA guidelines published by Growing Media Europe (2021) detail how to model and what to include for numerous substrates found in UA: peat, compost, coconut based materials, wood and bark materials, rockwool, and expanded perlite. Difficulties in including compost in UA LCAs were detailed in Section 3.4, and are also relevant for compost as a substrate material.

Limited details are available regarding lifetime and fate of permanent substrates in UA LCAs (i.e. not disposable ones for hydroponics). Dorr et al. (2017) evaluated a research-oriented rooftop farm that grew in substrate in raised beds, and assumed a 10-year lifetime of the farm, and that substrate had no end-of-life treatment as it would be donated and reused. In the end, the farm moved after about 10 years (because the university moved), and the substrate was donated to other urban farms (with great organizational effort). Kim et al. (2018) evaluated a rooftop farm and green roof, and assumed a 40-year lifetime based on the durability of the roof membrane material. Vacek et al. (2017) did an LCA of green roofs and assumed a lifetime of 20 years, noting that they would require renovation after this point. They assumed that substrate would be landfilled, and would be too degraded and unsuitable for recycling and reuse after the 20-year lifetime.

Recommendations:
Peat and peat moss have been well studied, and the processes available in LCA databases should be used. Impacts for coconut and wood/bark-based materials should be allocated on an economic basis between the main coconut and forestry products and the substrate byproducts (European Commission, 2010b). Any additional energy and water needed for processing the byproducts into substrate should be accounted for (Growing Media Europe, 2021). Residual waste products are those that have negligible economic value, and should only incur the impacts from their transport from the original site of use and their processing into a substrate (Growing Media Europe, 2021).

For permanent UA substrates (i.e. not disposable substrate in hydroponics and aeroponics), impacts from the substrate initially installed should be allocated over the lifetime of the farm or garden, similar to other pieces of infrastructure. This lifetime is usually highly uncertain, but a timeframe of 10-40 years can be considered. This value can be refined based on the orientation and precarity of the case study. Results can be sensitive to this assumption—

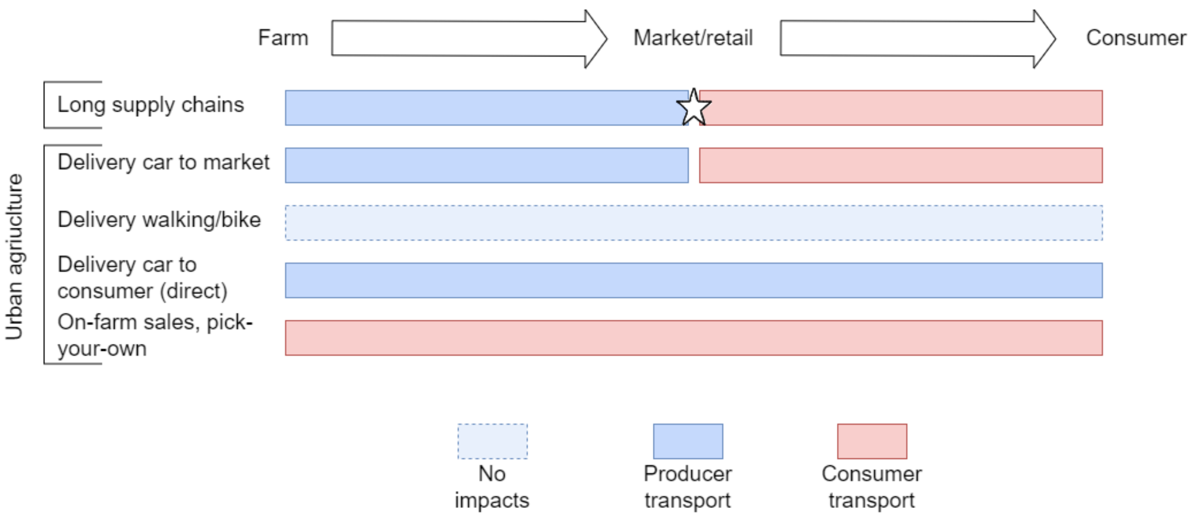

Figure 3.3 The downstream system boundary is shown for several simplified distribution schemes. Colored bars indicate who is doing the travel, and the empty bar for walking/bike indicates that there are no environmental impacts from this travel. Many rural food LCAs have a system boundary that ends at the market/retail stage (shown by the star). Several types of distribution networks, that are common for UA, are inconsistent with this system boundary because there is no equivalent market/retail stage. It is a simplified diagram because there could be additional steps between producers and consumers, and third-parties often do deliveries rather than producers themselves. UA LCA practitioners should ensure consistent system boundaries when comparing results with rural food LCAs

especially if substrate has a large contribution to impacts—so it is recommended to perform sensitivity analyses evaluating scenarios with different farm lifetimes. Disposable substrate used in hydroponics and aeroponics do not have the same lifetime considerations and can be treated as a regular input.

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

End-of-life for inorganic growing media will likely include municipal waste treatment or recycling. For organic growing media, the most common options are composting or field applications as a soil improver (Growing Media Europe, 2021). For composting, the farm/garden can be seen as the waste-generator described in section 3.3.4, and impacts of composting should be allocated between the waste-generator and the compost user. If substrate is applied as a soil improver by the next user, and no treatment or processing are necessary, then no impacts for waste treatment should be given to the farm/garden.

We recommend increased transparency and improved reporting regarding substrates in UA LCAs. The nature and the origins of substrate material should be clearly described, plus any physio-chemical characteristics, if available (Barrett et al., 2016). The amount of substrate initially applied, the amount added in amendments, the lifetime, and end of life waste treatment should be clearly stated. As with compost, the final fate of carbon is uncertain for organic materials in substrate, so if carbon sequestration is accounted for, it should be in separate results.

3.3.6 Transport and delivery

Challenge:

A main supposed environmental benefit of UA is its proximity of producers and consumers (Kulak et al., 2013; Weidner et al., 2019). Yet, knowledge is scarce about the transport and delivery of UA products—yet alone their environmental performance. This benefit is sometimes dismissed, considering that on average across all food products, transportation accounts for 6-11% of climate change impacts from food systems (Poore and Nemecek, 2018; Weber and Matthews, 2008). However, fruits and vegetables can have larger contributions to climate change impacts from transport (often 10-25%, but as high as 54%), due to the potential relatively lower impacts at the farm-stage, long distances, refrigerated transport, and airplane travel (Barbier et al., 2019; Bell and Horvath, 2020; Poore and Nemecek, 2018; Weber and Matthews, 2008). The benefit of reduced transport is mostly tested through comparisons of LCA results to the conventional long supply chains of rural agriculture, which many UA LCAs include (Dorr et al., 2021a). Challenges arise here in defining consistent system boundaries between urban and rural agriculture.

Post-farm transport in UA is often directly to the consumer. This is especially evident when products from UA are delivered by walking or by bike, because there are almost no impacts (very small impacts from street/sidewalk infrastructure and bicycle manufacturing, but these have been omitted in UA LCAs). In these cases, the system boundary implicitly includes the nil transport to the consumer (Sanyé-Mengual et al., 2018a, 2013). The final step of transport to/by the consumer, also called the 'last mile', is usually not included in food LCAs, and the system boundary ends at the market/retail stage (the star in Figure 3.3) (Pérez-Neira and Grollmus-Venegas, 2018). It is usually not included because it is difficult to model consumer transport behavior, and to isolate transport specifically for food purchases from other transport. Therefore, many comparisons between urban and rural agriculture products risk comparing a cradle-to-consumer UA system with cradle-to-market rural agriculture system. The last mile step is important because it can account for an even larger food transport distance (or 'food miles') than the transport over long food supply chains (Majewski et al., 2020). Impacts of the last mile for food (from customer travel) can contribute up to 21% of life cycle climate change impacts of pasta (Gnielka and Menzel, 2021), or 6% of urban food system climate change impacts (Stelwagen et al., 2021). Therefore, inaccurate comparisons here may omit a large benefit of reduced consumer transport of UA.

Another inconsistency that may arise in system boundaries between rural and UA is when consumers travel to the farm to purchase or harvest their own fresh produce. This can be referred to as direct sales, on-farm sales, U-pick, and pick-your-own, and is also seen in local food systems (which lack a common definition, but indicate geographic proximity between food producers and consumers). Research suggests that this travel of consumers to the farm accounts for large food miles, energy use, and climate change impacts, and are usually larger than a typical off-farm sales scheme, which delivers large amounts of product per trip (Coley et al., 2009; Enthoven and Van den Broeck, 2021; Majewski et al., 2020; Paciarotti and Torregiani, 2021; Schmutz et al., 2018). It is difficult to include this transport in LCAs, because of the system boundary, data about consumers' habits, and multiple purposes of trips (Christensen et al., 2018). Due to its potentially high impacts, it is important to further investigate customer travel to the farm in UA LCAs, although this would render results largely incompatible with the system boundaries of other food LCAs, which usually do not include travel to the grocery store.

Examples:
Transport from the farm/garden to the consumer on foot or by bike, or when production occurs in or on a building where consumers live or work, has been considered in several UA LCAs. They state that there are no processes or impacts for delivery (Figure 3.3) (Sanjuan-Delmás et al., 2018; Sanyé-Mengual et al., 2018a; Torres Pineda et al., 2020) (Chapter 2). Several UA LCAs include distribution by car to the consumer, based on a simplified model/distribution of transport modes and distances from the distribution point to consumers' homes (Hall et al., 2014; Hu et al., 2019; Pérez-Neira and Grollmus-Venegas, 2018). Other LCAs regarding urban food consumption and food products have focused on the last mile transport impacts (Bevilacqua et al., 2007; Melkonyan et al., 2020; Stelwagen et al., 2021).

Transport of the consumer for on-farm sales or pick-your-own has been measured in LCAs of local agriculture, but to our knowledge, our case studies were the first to include this for UA (Chapter 4.1). Other purposes of the trip can be considered, where for example if the customer is already traveling for another purpose, then impacts of the trip can be allocated between the purposes (Majewski et al., 2020; Mundler and Rumpus, 2012). Our UA case studies showed variable increases in climate change impacts when we included this (14-78% increases). A distinction between UA and local agriculture here is that in the urban setting, we can expect customers to travel more by walking, bicycling, or on public transport, which can have less impacts than the car-dominating local-food examples (Chapter 4.1).

Recommendations:
We recommend that in general, UA LCAs include post-farm delivery processes to account for the unique urban position (Weidner et al., 2019). Since there may be large uncertainties in delivery logistics, and inconsistent system boundaries with rural systems, results should be presented with and without post-farm transport, giving cradle-to-farm gate and cradle-to-consumer or market impacts (Sanyé-Mengual et al., 2015a). This is particularly relevant for comparisons to rural agriculture, because proximity to the consumer is a core characteristic and environmental benefit of UA. The delivery scheme of a case study should be clearly described, including the transport distances, modes, and frequencies of deliveries. Impacts from delivery should be specified (i.e. not only grouped into a post-production category) in order to highlight this unique aspect of UA.

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

It is difficult to make a generalized recommendation for including consumer transport to the farm. On one hand, it is important to include post-farm steps in UA LCAs, and the decision to have customers travel to the farm is a system management decision by farmers. On the other hand, consumer transport to grocery stores is commonly considered outside the scope of food

LCAs because it is determined by individual consumer decisions, and consumers may be making those trips regardless of the system studied (Stelwagen et al., 2021). Ultimately, its inclusion likely depends on the goal of the study and the data reliability. If customer transport to the farm is included, it should be presented as an additional set of results, due to the likely high uncertainty. Data about customer trips may be difficult to collect, but should include mode of transport, distance, and other purpose/s of the trip. Such preliminary work would be essential for evaluating the effects of scaling-up UA on urban transport logistics (Oliveira et al., 2021). Research focused on this topic may aim to identify tipping-point distances where customer transport to the farm causes impacts greater than those of conventional supply chain distribution (Coley et al., 2009; Sanyé-Mengual et al., 2015a), or evaluate the effect of such transport at the city or regional scale. Variability should be accounted for, using techniques described in the following section (section 3.7). Such studies may be particularly complex and valuable to perform in an urban context, considering additional possible modes of transport such as public transit.

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

3.3.7 Variability and uncertainty of UA

Challenge:

Variability refers to real and essential differences, as opposed to uncertainty which comes from choices, simplification, and lack of data (Hauck et al., 2014). Uncertainty can be reduced with additional data collection, which may be particularly challenging for UA (section 3.3.3). Variability is inherent in systems and cannot be eliminated without major changes to the goal and scope. Agricultural LCAs have particular issues with high variability because of diversity in controlled factors like farming practices and logistics, and in ‘natural’ factors like climate and soil characteristics (Lam et al., 2021a; Mclaren, 2010; Notarnicola et al., 2017). Here, we hypothesize that the controlled factors are even more variable in UA than in rural agriculture, due to the physical system, human and experience elements, and the novel and (sometimes) unprofessional context. These lead to less standardized and predictable systems (Christensen et al., 2018)(Chapter 1).

The diversity of physical forms that UA can take (on the roof or on ground, in the soil or in raised beds in substrate, for consumers in the building or with complex intra-urban delivery) suggests that a diversity of practices would emerge. The urban setting introduces physical limitations which spur diverse outcomes in growing practices, including shading from buildings, poor-quality anthropogenic soils, air pollution, and limited access to materials (Taylor, 2020; Wagstaff and Wortman, 2015). Human elements such as motivation for urban farming and gardening, years of experience, and access to agronomic information and training are highly variable, and likely affect growing practices (McClintock et al., 2016; Taylor, 2020). More broadly, the novel and unprofessional status of much of UA means that it has not converged towards optimized, standardized operations. In contrast, rural agriculture has been researched for decades, and is relatively consistent due to knowledge, experience, training, university agricultural extensions, and technology such as tractors, crop varieties, and chemical inputs (Armanda et al., 2019; O'Sullivan et al., 2019).

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

This also leads to variability in UA overall (i.e. between systems). Indeed, in the review of UA LCAs (Chapter 2), we noted that there were few actual replicates of systems due to the numerous variables (growing technology, motivation, climate, and many others not described in the UA LCA literature), which made it difficult to compare results. This poses a challenge to understanding the general performance of UA, since there is not really a 'general' situation for UA.

Examples:
One of the most common ways of addressing variability and uncertainty in UA LCAs is presenting alternative scenarios in the form of sensitivity analyses, to model impacts if a different parameter or setup was chosen. This is done to model different infrastructure lifetimes (Dorr et al., 2017; Martin and Molin, 2019), yield (Romeo et al., 2018; Rufí-Salís et al., 2020b), light efficiency for indoor systems (Pennisi et al., 2019; Shiina et al., 2011), use of raw or reused materials for infrastructure (F. Corcelli et al., 2019; Sanyé-Mengual et al., 2015b), emission/leaching rates of fertilizers and pesticides (Rothwell et al., 2016), and amount of inputs used like water, electricity, and fertilizers (F. Corcelli et al., 2019; Romeo et al., 2018). Another strategy was to use ranges of inventory values, which give ranges of results. This has been done for delivery/distribution schemes (Hu et al., 2019; Pérez-Neira and Grollmus-Venegas, 2018; Stelwagen et al., 2021) and water demand (Caputo et al., 2020). When parameters with high variability are identified, the goal of the LCA can shift to find tipping points where one system performs better/worse than another (usually UA vs rural). This was done for yield and distance from producer to consumer (Kulak et al., 2013; Sanyé-Mengual et al., 2015a). We used Monte Carlo simulations in our case studies (Chapter 4.1) to quantify ranges of results based on distribution of composting parameters.

Recommendations:
Variability and uncertainty within systems can be reduced or accounted for with several strategies. Temporal variability, due to annual climate differences or changes in operations (for example due to farmer turnover), should be reduced by collecting data for multiple years and using an average of values, or selecting the most representative year (Loiseau et al., 2020). Specialized indicators can be used that quantify how important variability is for a system (Hauck et al., 2014). Variability of inventory items should be accounted for using distributions or ranges (Stelwagen et al., 2021), and probabilistic simulations, such as Monte Carlo simulations (Huijbregts, 1998). When a parameter has high uncertainty and large effects on the final result, the goal of the LCA may be modified to determine values of that parameter that improve performance or represent tipping points (Loiseau et al., 2020).

Variability between systems is problematic when trying to compare or summarize results for similar systems. Such comparisons are necessary to draw trends and generalize LCA findings, which is a feature of rather mature LCA research topics. Fewer technical recommendations can be made here, but we note that more holistic descriptions of case studies would help make sense of the body of literature. Plus, simply increasing the number of UA LCAs would allow for more meaningful statistical tests and more reliable average values for certain systems (Huijbregts, 1998).

3.4 Research directions for UA LCAs

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

3.4.1 Align with urban land uses and green infrastructure LCAs

Presentation:

The UA LCA literature is dominated by a product-based perspective, which inherently places the focus on the food-production function of UA. UA distinguishes both the unique, non-rural position of agriculture, plus the non-conventional use of urban space (Neilson and Rickards, 2017). The latter perspective has not been widely studied with LCA, except for studies comparing different uses of rooftops for flower gardening, farming, or solar panels (F. Corcelli et al., 2019; Goldstein et al., 2016b; Kim et al., 2018). Furthermore, many researchers and participants claim that the main focus of UA is social and recreational purposes, which could be found in other urban land uses, and food production is a welcome and non-negligible side product (Kirby et al., 2021; Sanyé-Mengual et al., 2020). Similarly, UA is an option for urban green infrastructure, among many others. In this case, UA may be more comparable to a park or other social/recreational activity than it is to rural agriculture. There is a wealth of literature on environmental assessments of green roofs (Kim et al., 2018), urban parks and forests (Strohbach et al., 2012), golf courses (Tidåker et al., 2017), urban wetlands (Duan et al., 2011), grassy areas (Smetana and Crittenden, 2014), and other green infrastructure (Nicese et al., 2021), and it would be useful for UA LCA practitioners to relate UA to these land uses. It could provide meaningful comparisons to similar systems, and illuminate shortcomings in UA LCAs that have not emerged due to the so-far limited product-based perspective. For example, urban green infrastructure LCAs found that waste treatment of biomass can be highly impactful (Nicese et al., 2021; Tidåker et al., 2017), and results can be highly sensitive to carbon sequestration (Strohbach et al., 2012; Tidåker et al., 2017), which has not emerged in UA LCAs.

Recommendations:

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

3.4.2 City-scale/Scaling up

Presentation:

In addition to the food-production and land-use perspective, UA is framed in the context of sustainable cities (Petit-Boix et al., 2017). Evaluating the effects of UA on resource consumption, food provisioning, and environmental impacts at the city-scale is useful to determine the relative magnitude of findings from the farm-scale. It is also useful to identify emergent processes at the city-scale, which are not evident at the farm/garden scale, such as effects on municipal organic waste treatment or urban transport logistics.

Researchers have modeled the effects on the city of “scaling up” or developing UA under different scenarios. Goldstein et al. (2017b) evaluated the effect of installing UA in available land in Boston, USA, and found that it could reduce food-related climate change impacts at the city level by 1-3%, and increase land occupation by 1%. Mohareb et al. (2018) performed a similar analysis for the USA and found food sector greenhouse gas emissions were reduced by 1%. Other scaling-up analyses suggest that UA could ‘absorb’ and compost 9% of municipal organic waste in Boston (Goldstein et al., 2017b), and 17% and 52% in Lyon, France and Glasgow, Scotland (Weidner and Yang, 2020). In a further step, researchers have performed LCAs on the measured or estimated consequences of implementing UA, such as through changes in diets and food waste behaviors, but do not necessarily account for the impacts of operating UA (Cleveland et al., 2017; Puigdueta et al., 2021). Extrapolating farm-level results to the city-scale helps provide perspective, because if fruits and vegetables are substantially more or less impactful than rural products, but at the city or individual diet scale they are a drop in the bucket, then maybe the framing of UA LCAs needs to shift.

UA is embedded in the infrastructure and functioning of specific cities, which provide certain environmental constraints or opportunities based on the city context (Martin et al., 2016). For UA LCAs, some characteristics of the specific city are inextricably included in the LCA results. For example, a well-known factor at the country level is the electricity grid, and studies have shown that when modeling the same case study using the electric grid of different countries, impacts can vary by up to factor of 8 (Dorr et al., 2021a). Similar factors at the city-level may influence UA environmental performance, such as city density (Montealegre et al., 2021). High-density cities may have limited available space for ground-based UA, and as a result rooftop UA may be more prevalent. Rooftop UA may require building reinforcement, which can have large environmental impacts (Goldstein et al., 2016b), but also provides the opportunity for integration with waste flows from buildings (like for water, heat, and CO_2) (Sanjuan-Delmás et al., 2018). Density may also affect the type of delivery method, where high-density cities have a greater possibility for delivery on foot or bicycle. The building stock in a city may affect UA’s form and impacts: for example, older buildings are more likely to need structural reinforcement for rooftop UA (Ledesma et al., 2020b), and flat roofs are much more suitable than slanted roofs (Weidner and Yang, 2020). Availability of public transportation may affect impacts of customers travel to farms or gardens. The typical waste treatment scheme for organic waste in a city would largely influence the potential for avoided burdens related to compost—i.e. if organic waste is composted anyway through the city. Finally, the benefits of reduced food miles for rural products are context-specific, and depend on the actual source and distribution network of products to a city (Bell and Horvath, 2020; Edwards-Jones et al., 2008; Hospido et al., 2009).

Recommendations:
LCAs at the farm level should account for the urban context with clearer descriptions of the city. This can include characteristics such as the position of the farm in relation to the city

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

3.4.3 Ecosystem services and positive impacts

Presentation:

LCA is inherently poised to evaluate the negative (adverse) impacts of a system rather than its positive impacts (benefits). The ecosystem service (ES) concept takes the opposite perspective, defined as the benefits that people obtain from ecosystems (Millennium Ecosystem Assessment, 2005). ES assessments are better poised at measuring some aspects of UA than LCA, and combining the two ways of thinking would allow for more comprehensive assessments of UA and other urban green infrastructure (Romanovska, 2019). There is no consensus on an ES measurement framework (Maia de Souza et al., 2018), although there are many tools and frameworks available (Grêt-Regamey et al. (2017) evaluated 68 of them). Much work has been dedicated to the consideration of ES in LCA (Maia de Souza et al., 2018; Othoniel et al., 2016; Tang et al., 2018; Zhang et al., 2010), although no method is consistently used. UA LCAs so far have not integrated ES, as some rural agriculture LCAs have done through allocation between ES (Boone et al., 2019) or with ES modeling (Chaplin-Kramer et al., 2017). ES may be fully integrated into the LCA methodology (i.e. with additional impact pathways for LCA, or integrating the cascade framework of ES into LCA), or may be more loosely integrated though qualitative or quantitative interpretation of results calculated separately from an LCA (De Luca Peña et al., 2022). Full integration of ES in UA LCA may not be currently operational, but UA is a particularly rich topic through which to promote methodological development of ES and LCA, and would offer useful case studies for future research.

ES have been widely measured as a benefit of UA (Artmann and Sartison, 2018). Their perceived or potential benefits are often qualitatively evaluated through interviews with stakeholders and ranking of ES (Aerts et al., 2016; Camps-Calvet et al., 2016; Sanyé-Mengual et al., 2018d, 2020) or quantitatively measured with indicators (Cabral et al., 2017; Grard et al., 2018). There are four types of ES: provisioning (i.e. food production), regulating (i.e. stormwater runoff regulation) cultural (i.e. recreation) and supporting (i.e. pollination) (Millennium Ecosystem Assessment, 2005).

The most apparent provisioning ES of UA is food production. As many UA LCAs use a FU based on food production, they essentially quantify the impact of this ES. Boone et al. (2019) demonstrated a method to allocate between this provisioning ES of agriculture and other ES in an LCA, which highlighted that food was not the only ES (or ‘output’) of agriculture.

Some regulating ES of UA that have been measured include water runoff regulation, organic waste recycling, and microclimate regulation (Dennis and James, 2017; Grard et al., 2018). Benefits of avoided stormwater runoff have been quantified with LCA, and offset 13-72% of several impact categories (Goldstein et al., 2016b; Kim et al., 2018). Carbon sequestration can also be evaluated using LCA or ES (Orsini et al., 2014), and its implication in LCA is described in section 3.3.4.4. Reduction of the urban heat island effect is a frequently proposed regulation ES of UA, and is generally excluded from all LCAs (Susca and Pomponi, 2020).

Cultural ES are sometimes perceived as the top benefit of UA, and include recreation, beautification, cultural identity, social cohesion, community building, education, and health (Giacchè et al., 2021; Sanyé-Mengual et al., 2018c). Indicators to measure cultural ES at the farm/garden level include the volunteer hours, number of educational and recreational activities offered, and their number of participants; and at an individual level may include types of skills acquired, number of new people met at the site, or survey responses on a numbered scale of physical and mental impacts (Dennis and James, 2017; Giacchè et al., 2021). Cultural ES may provide a framework to include social benefits in UA LCA assessment (detailed more in section 3.4.4).

The role of biodiversity in ES is foundational, as it is defined as the source of ES (McDonald et al., 2013; Millennium Ecosystem Assessment, 2005). As such, biodiversity is often used as a proxy indicator for ES (specifically supporting ES) (Cabral et al., 2017). Improved local biodiversity is perceived as an important environmental benefit of UA (Camps-Calvet et al., 2016; Sanyé-Mengual et al., 2018c) and is frequently measured in the context of ES of UA (Dennis and James, 2017; Quistberg et al., 2016; Woods et al., 2016). This benefit is not accounted for in LCA. Biodiversity impacts in LCA have been the subject of methodological development for decades, and it is usually framed as the impact *on* biodiversity *from* land use (or other ecological damage, although most frequently land use) (Teixeira et al., 2016). LCA models the upstream and downstream impacts of materials and processes on biodiversity around the world, and is especially useful at detecting biodiversity impact hotspots in supply chains (Teixeira et al., 2016). Local biodiversity is not considered, so other methods are more relevant for farm-scale biodiversity impacts (Frischknecht et al., 2016). This can be measured using metrics and indicators like species richness, habitat fragmentation, habitat vulnerability, or land use intensity indicators (Frischknecht et al., 2016; Pepin, 2022).

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

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

3.4.4 Social benefits and life cycle sustainability assessment

Presentation:
A main strength of UA is its multifunctionality, with important social functions (Gomez Villarino et al., 2021; Orsini et al., 2020; Pourias et al., 2016). This isn't reflected in UA LCAs, but it should be, since core principles of LCA are evaluating the main function of a system (through selection of a FU), and accounting for multiple outputs (through allocation and system expansion).

Accounting for social aspects of an activity is a main issue for LCA, and social LCA (S-LCA) is a promising yet nascent strategy to overcome this (UNEP/SETAC, 2009; Zimek et al., 2019). Using life-cycle thinking, S-LCA tracks the social impacts of a product through its life cycle. S-LCA is oriented towards quantifying negative impacts, and therefore may not be appropriate for evaluating the social aspects of UA, which are generally considered to be beneficial. S-LCA databases offer data for social impacts embedded along the supply chain, but the information necessary for UA is more relevant at the farm/garden, neighborhood, or city scale (Romanovska, 2019). Plus, such databases are not as generalizable as large LCA databases. A strength of S-LCA is its ability to account for the perspectives of multiple stakeholders, such as workers, consumers, and the local community. This is especially useful to evaluate the potential for UA to address social justice issues, by highlighting not just which social benefits are brought, but who they are affecting. S-LCA currently lacks agreed upon social indicators, with more than 150 recorded in use (Finkbeiner et al., 2010). This is because they are usually defined through stakeholder engagement, making consistent methods and comparisons between studies difficult (Fauzi et al., 2019). Peri et al. (2010) outlined indicators for S-LCA of green roofs, including area of green roof made accessible to the public, fair salary, working hours, air pollutant levels, outside air temperature, and agricultural production.

Apart from S-LCA, an option to include social benefits of UA may be to consider its multifunctionality with traditional LCA practices. For example, allocation can be used to distribute impacts based on relative importance of food production vs social benefits. This allocation may be done based on the level of ES provided by each activity, as done in Boone et al. (2019). Alternatively, it may be based on the relative sources of revenue from food sales vs grants vs other activities. The other main method for dealing with multifunctionality in LCA is system expansion, which is probably not an option here, because it is difficult to identify and quantify the alternative sources of social benefits that UA provides. If social goals are the main function of a farm, we can imagine using a FU based on the social "output", such as volunteer hours or total number of new people met by UA participants. These methods may be unconventional, and difficult to compare to existing studies, but would be interesting to explore.

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

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

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

3.5 Summary of key points

Here we summarize the key recommendations for performing UA LCAs, which sometimes intersect the research directions.

1. Be transparent, thorough, and critical when evaluating compost, substrate, and other organic inputs. They are especially important for UA, and are not usually the focus in agricultural LCAs.
2. Use sensitivity analyses for important parameters with high uncertainty or variability to obtain a range or distribution of results. Such parameters may be related to:
    - Infrastructure lifetime
    - Substrate lifetime
    - Compost emission factors
    - Delivery logistics
    - Customer travel to the farm
3. Present results with and without major avoided burdens and carbon sequestration benefits.
4. Use multiple FUs—at least land and product-based.
5. Provide more holistic descriptions of UA case studies, because UA is diverse and vaguely defined.
6. Describe the representativeness, scale of production, or innovative status of a case study.
7. Include post-farm transport of products. If this is done by bike or on foot, and processes are not included because there are no impacts, the system boundary should be considered cradle-to-consumer. Provide separate post-farm impacts to allow for harmonization of system boundaries with other systems.
8. Work with more functioning case studies to collect primary data, because UA may not operate as expected or as measured under ideal, controlled conditions.
9. Compare impacts with an area-based FU to other urban green infrastructure.
10. Consider the effect of the city and local context on the performance of UA, as is frequently done with electricity grids.
11. For more precise comparisons to rural agriculture, seasonality and local context should be considered.
12. Include social, economic, and ecosystem service-related measures, even if they are not life-cycle based.

3.6 Conclusion

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

Chapter Four: Life cycle assessment of urban agriculture case studies: application of a novel methodological framework

4 Chapter Four: Life cycle assessment of urban agriculture case studies: application of a novel methodological framework

This is the only chapter of the dissertation that is composed of two parts. Both parts include original life cycle assessments of urban farms and gardens. Part 1 covers eight urban farms and community gardens in Paris, France and the Bay Area, California, USA. Part 2 covers an urban mushroom farm in the Paris area.

4.1 Part 1: Life cycle assessment of eight diverse urban farms and community gardens

This part presents the life cycle assessment of eight urban farms and community gardens in Paris, France and the Bay Area, California, USA. It has been prepared for publication in the journal Science of the Total Environment, with an expected submission in May 2022.

Life cycle assessment of eight diverse urban farms and community gardens

Abstract

A common theme in the claimed benefits of urban agriculture (UA) is in the environmental dimension. UA may support urban biodiversity, alleviate stormwater runoff, and produce vegetables and fruits with low embodied impacts thanks to virtuous growing practices and a hyper-local position. Such embodied impacts are usually measured with life cycle assessment (LCA), a method that models multiple environmental impacts of a product or service over all stages of its life. LCAs of UA have emerged relatively recently, and show mixed evidence regarding UA's environmental performance. In a parallel work, we created a methodological framework to bring consistency and completeness to UA LCAs. Here, we present the UA LCA case studies that simultaneously informed and demonstrate this framework. We worked with eight urban farms and community gardens in Paris, France and the Bay Area, California, USA, and collected primary data from one year, and performed thorough LCAs according to the framework. The case studies represented diverse growing systems, ranging from low-tech sites growing in the soil or in raised beds, to medium-tech sites with open-air hydroponics and strictly managed vertical growing structures. They also covered multiple orientations of UA,

with commercial and non-profit systems, and motivations ranging from education for students, community building, civic engagement, and commercial systems. Our research questions were, how does environmental performance vary by the geographic location and motivational context of UA? And, what are the sources of variation in environmental performance of UA? We found that rankings in environmental performance depended on whether we used a functional unit based on product (kilogram of crop) or area (m^2 food growing area). The more professional and medium-tech farms, which were mostly found in Paris, had lower impacts using a kilogram-based functional unit, but the social-oriented farms and gardens (mostly in the Bay Area) had lower impacts with an area-based functional unit. Large potential impacts came from infrastructure, irrigation, compost, and peat for seedlings. Despite the diversity in the systems studied, our results had lower variability than what is found in the literature for UA LCAs, thanks to the consistent modeling and data choices that came from applying the methodological framework.

4.1.1 Introduction

Urban agriculture (UA) is the growing of food in and around cities, and is becoming more prevalent in practice and in research (Mok et al., 2013; Pinheiro et al., 2020). In the Global North, UA is recognized as a mostly multifunctional activity, where growing food is one of several objectives and benefits (Orsini et al., 2020). Other objectives and benefits of UA include education, community development, improving the environment, recreation, climate change mitigation, improving urban biodiversity, and organic waste recycling (Kirby et al., 2021; Siegner et al., 2020; Tuijl et al., 2018; Weidner et al., 2019). Still, the agricultural function remains a top priority among stakeholders in the context of food security, food justice, commercial operations, and access to fresh produce (Kirby et al., 2021; Pourias et al., 2016; Siegner et al., 2020). It is well known that agriculture drives many environmental issues, such as climate change, water depletion, energy use, environmental pollution, and biodiversity loss (Campbell et al., 2017). The contribution of UA to these issues is gaining attention, especially as researchers and local leaders call for ‘scaling up’ UA in cities and must avoid promoting an activity with large environmental burdens (Armanda et al., 2019; Mohareb et al., 2017).

Life cycle assessment (LCA) has proven useful to evaluate these environmental issues for rural agriculture. LCA is a standardized method that models environmental impacts of a product or service throughout its life cycle, from “cradle to grave” (ISO 14040, 2006). Due to some unique characteristics of agriculture, methodological adaptations were necessary to improve the relevance of LCA (typically used for industrial processes) in this application (Audsley et al., 1997; Caffrey and Veal, 2013). These unique characteristics include biological and ecological dynamics, high uncertainty, seasonality, non-point source emissions, a distinct set of inputs with initially unknown inventories (such as fertilizers and seedlings), and new system modeling issues around co-products (Nemecek and Gaillard, 2010; Notarnicola et al., 2017). A relatively large number of LCAs have been done for conventional, open-field agricultural crop production, with findings that converge across the whole body of literature. This has allowed for some generalizations regarding impactful processes, typical ranges of values, and relative performance of different farming methods (Parajuli et al., 2019; Poore and Nemecek, 2018; Seufert and Ramankutty, 2017).

Such convergence and knowledge creation have not been achieved yet for UA, for which application of LCA is in its infancy. In a recent review and meta-analysis, we showed that the available UA LCAs use variable perspectives and system modeling decisions (Chapter 2) (Dorr et al., 2021a). It was difficult to draw generalizations from the literature due to high variability in results and methods, small sample size, lack of primary data from urban farms and gardens, inconsistency in what was included, and inconsistent reporting (such as different processes included in life cycle stages). Typical ranges for climate change impacts were difficult to summarize due to their large variability, and other measures including water and energy use were scarce, but also highly variable. In response, we developed a framework (Chapter 3) to guide LCA practitioners to perform more holistic, consistent LCAs of UA. This framework was created through an iterative process, where the framework was informed by work with case studies (presented here), and the case studies here adhered to the framework.

In this paper, we present LCAs of eight diverse urban farms and gardens in two geographies: Paris, France and the Bay Area, California, USA. The objectives were to 1) simultaneously inform and demonstrate the framework outlined in Chapter 3, and 2) perform comprehensive, consistent, and rigorous LCAs of diverse UA, based on primary data, to contribute to the knowledge around its environmental performance. Regarding the second goal, we had two research questions: how does environmental performance vary by the geographic location and motivational context of UA? And, what are the sources of variation in environmental performance of UA? These geographies were chosen because of their different population densities, climate, and context of UA (i.e. its history and main orientation). Plus, UA is prevalent in both locations, with interest from local researchers, governments, and practitioners (APUR, 2017; Glowa, 2014).

4.1.2 Methods

Here we describe the case study farms and gardens, the selection process of farms and gardens, data collection, and finally the LCA method, including goal and scope definition, life cycle inventory, and impact analyses. More details about our data collection and data sources are available in the Appendix.

4.1.2.1 *Case study cities*

The case studies were located in Paris, France and the Bay Area, California, USA. We expected that covering UA in varied backgrounds would allow different aspects of UA to emerge, similar to studying diverse farms and gardens.

Four case studies were assessed in Paris, France, or the cities on the border of Paris. Paris had a population density of 20,913 inhabitants/km^2 in 2020 (OECD, 2021), and a climate categorized as temperate oceanic (Cfb Köppen-Geiger classification) (Beck et al., 2018). Although UA has historically been present in Paris, there is a recently renewed interest. Traditional forms of UA include allotment and community gardens, with farming in and on buildings developing only within the last five years approximately (Demailly and Darly, 2017). It is often done in the context of access to green space and nature, community development, education, and professional/commercial reasons (although of course there are many examples of UA in the city with other objectives) (ADEME, 2017; Demailly and Darly, 2017). The city promotes UA through programs that support its development such as Parisculteurs (www.partisculteurs.paris). Due to the high density of the city, rooftop UA is increasingly relevant in the recent wave of UA projects (Demailly and Darly, 2017).

The other four case studies were located in the Bay Area, California. The Bay Area is a region that includes cities such as San Francisco, Oakland, Berkeley, and San Jose. Specifically, we worked with case studies in San Francisco, Berkeley, and El Sobrante. Here, UA has a longer and steadier history, and is related to issues of food justice, food access, and education, among many others (Bradley and Galt, 2014; McClintock, 2011; Siegner et al., 2020). The density for the region (San Jose, San Francisco, Oakland) was 118 inhabitants/km^2 in 2020 (OECD, 2021). For only San Francisco this was 7,223 inhabitants/km^2, highlighting the heterogeneity and prevalence of more suburban neighborhoods in the region (U.S. Census Bureau, n.d.). The climate is classified as warm-summer Mediterranean (Csb Köppen-Geiger classification) (Beck et al., 2018), but important microclimates mean that San Francisco is cloudier and colder than Berkeley, and even more so than El Sobrante (Gilliam, 2001). Most UA here is ground-based.

4.1.2.2 *Farm and garden selection*

We aimed to study a diverse selection of UA, in regards to technical systems, location, and motivations. While our case studies were indeed diverse, they were selected as the result of convenience sampling. Ultimately, we could only work with sites where the farmers/gardeners were interested and willing to participate. Because collecting data for an LCA is a time-consuming task, it was not simple to find farmers/gardeners willing to participate. The sites in Paris were identified through existing relationships between the researchers and farmers. The sites in California were identified through an announcement in a regional UA newsletter, requesting that interested farmers/gardeners contact us. In addition to the eight farms/gardens studied here, we also began work with five others that eventually didn’t work out due to farmers’/gardeners’ lack of time and modifications in the technical system during the study. We especially had difficulty finding partners at indoor hydroponics farms, representing ‘vertical farming’ or ‘plant factories’, and ultimately were unable to include any such case studies.

4.1.2.3 *Description of the case studies*

Most sites preferred to remain anonymous (especially commercial ones), so all farms are named based on their location (FR and US) and a number. Although they had some concerns about sharing detailed data (such as names of products used or mixes of fertilizers), researchers had full access to this data. Some main characteristics of the farms are in Table 4.1, showing descriptions of the farms/gardens based on their general attributes, plus quantitative data that were collected during this study. Food production, water use, and compost use data refer to continuous 12-month periods between 2019-2021. We worked with both commercial (FR1, FR2, US3) and non-commercial UA (FR3, FR4, US1, US2, and US4). Note that non-commercial sites still sold produce, but it was not their main purpose or source of funding. Similarly, commercial sites also had non-monetary goals, especially US3, with a main goal of education as a pick-your-own farm where customers come and learn about the crops grown. Non-commercial sites had goals of education, job training, community building, and research. It was difficult to assign main goals to farms because they often had several main goals, and certainly provided other services even if they were not a main goal, illustrating the multifunctional nature of UA. Typically, for UA, “farm” indicates a commercial site and “garden” denotes a non-commercial site (Reynolds and Darly, 2018). For brevity, we refer all sites as farms in the rest of this chapter.

4.1.2.4 *Management structure*

The farms also differed in who and how many people did the farming and decision making. FR4 and US4 were school gardens, with one main farm manager and much of the work done by middle and high school students. US2 was similar in that one person led the site, and work was mostly done by university students and volunteers, but its main objective was research. FR3 and US3 had mostly centralized work forces of several experienced farmers, with rotating trainees or interns. FR1 and FR2 were farmed only by a handful of experienced farmers. US1 was a community farm with many community members with varying levels of expertise working on the farm, and had a decentralized management and decision-making system, with no main farm manager.

4.1.2.5 *Physical setup*

There were a variety of technical farming methods and physical set-ups. The four Paris farms were all on rooftops, and the California farms were all ground-based. This was likely a result of the high density and lack of space in Paris compared to the Bay Area (although ground-based UA can be found in Paris as well, and vice versa). FR1 and FR2 can be considered ‘medium-tech’—as opposed to ‘high-tech’ UA which consists of indoor vertical systems, usually with artificial light and temperature regulation (Orsini et al., 2020). FR1 grows in vertical structures filled with substrate with crops growing out of the sides and tops, providing a large growing surface area per ground area (note that the ground area was used for the food growing area in analyses, not the surface area of the vertical facades). FR2’s system is outdoors on a roof and consists of two subsystems: hydroponics, and aeroponics (where plant roots are exposed to the air and misted with fertilizer and water). FR3 grew on the roof in a monolith layer of substrate covering the entire roof, with no need for raised beds, while FR4 grew on the roof in raised beds. US1 and US2 grew in the soil in spaces historically used for agricultural research. US1 also had a large medicinal herb area and children’s garden. US3 built up a soil over time on an unfavorable surface, and has a small and dense main vegetable area, an orchard, and other productive trees and blackberry brambles scattered throughout the site. US4 was established on top of an asphalt athletics field of a school, and has been slowly building up soil with compost applications over the years, but never had a large application to ‘create’ the substrate. All farms except FR1 and FR2 used drip irrigation controlled by timers, supplemented with watering by hose when necessary. FR1 used drip fertigation. FR2 used fertigation supplied through spraying roots in the aeroponics system, and flowing through roots in the hydroponics system.

4.1.2.6 *Farming practices and inputs*

All farms except for FR1 and FR2 used compost as the main input. Several farms also used small amounts of organic fertilizers such as feather meal, kelp meal, ground oyster shells, mushroom compost, and straw. Pest control did not seem to be a major concern at most farms: only one farm reported using an organic iron phosphate-based slug killer, and another released predatory insect

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		FR1	FR2	FR3	FR4	US1	US2	US3	US4
Description	City	Rosny sous Bois	Paris	Aubervillier s	France	Berkeley	Berkeley	El Sobrante	San Francisco
	Position	Rooftop substrate vertical	Rooftop. Hydroponic aeroponic	Rooftop substrate	Rooftop substrate	Ground in soil	Ground in soil	Ground in built up soil	Ground in built up soil
	Main goal(s)	Commercial , food production	Commercial , food production	Job training, food production	Education	Community building, education	Research, food production	Commercial , education*	Education
	Year of establishment	2019	2020	2016	2018	2012 historically agricultural land	historically agricultural land	2012	2004
	Degree of social engagement	Low	Low	Low	High	High	Medium	Medium	High
Area	Total farm area (m^2)	2600	1490	700	1791	6336	854	3541	2390
	Food green area (m^2)	253*	298	397	248	880	610	635	554
Food	Annual harvest (kg)	6924	7999	1771	475	2117	741	922	312
	Yield (kg/m^2)	27.4	26.8	4.46	1.92	2.41	1.21	1.45	0.56

	Number of crops	23	18	36	39	47	14	129	19
Water	Annual water use (m^3)	1644	1920	2078	213	2035	375	1080	819
	Water use by food (m^3/kg)	0.24	0.24**	1.17	0.45	0.96	0.51	1.17	2.63
	Water use by area (m^3/m^2)	6.50	6.44	5.23	0.78	2.01	0.61	1.14	1.34
Compost	Amount compost used (m^3)	0.00	0.00	2.00	7.90	15.61	11.30	16.80	12.02
	Compost (kg/m^2)	0.00	0.00	3.02	17.33	9.24	11.11	10.62	12.13
	Compost per kg (L/kg)	0.00	0.00	1.13	16.63	7.37	15.25	18.22	38.55

Table 4.1 Food production, water use, and compost use data refer to continuous 12-month periods between 2019-2021. Degree of social engagement was assigned by researchers. Low engagement farms were not usually open to the public or did not hold events that brought in the public, and few people (mostly employees) did the farming. Medium engagement farms welcomed specific outside groups, usually students, and farming was done mostly by employees and with the help of volunteers. High engagement farms encouraged participation from the public and were farmed roughly equally by both employees and volunteers. *FR1 grows in vertical structures. This area refers to the ground area covered by those structures, not the surface area of the facades. **FR2 had no data available regarding water use. We used data from their records, rather than launching a dedicated data collection campaign. We assigned the same water use per m^2 as FR1, since they also used precise, low-consumption drip irrigation in vertical structures.

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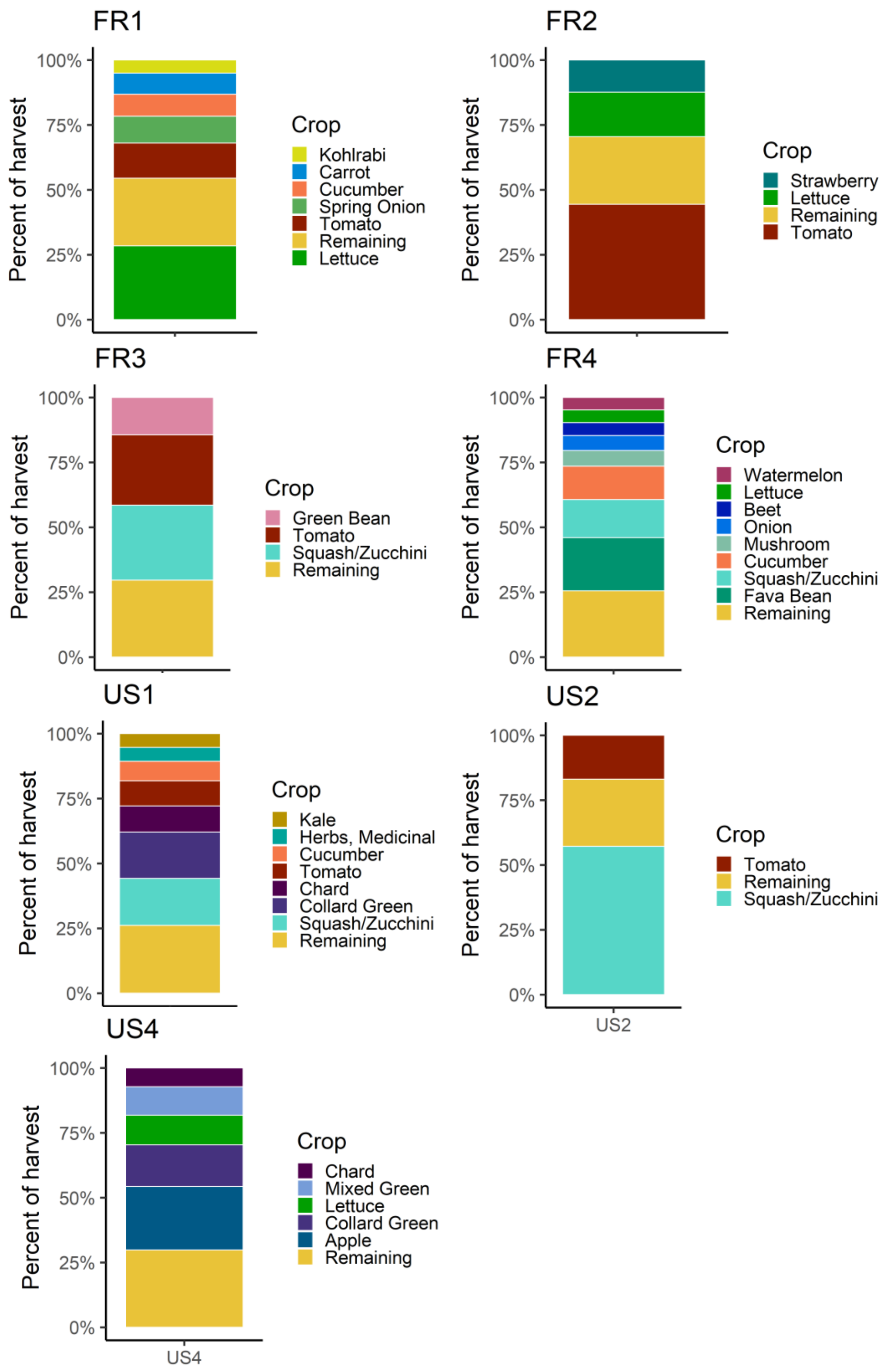

Figure 4.1 The annual harvest of each farm was broken down by crop type, for the crops that made up 75% of the harvest at each farm. The remaining harvest was grouped into one category for each farm, labeled "Remaining". US3 was excluded because they only recorded total harvest, rather than harvest per crop.

4.1.2.7 *Crop selection*

Farms usually adhered to a crop plan created at the beginning of each year. Some farms were more efficient than others at harvesting crops once they were ready, and replanting with new crops. For example, at US1 and US3, plants were often left in the ground to flower because gardeners do not manage to replant them quickly, and because they want to leave flowers for pollinators. Many farms dedicated space and inputs to ornamental or pollinator-attracting flowers and shrubs. We excluded this from the study where it was possible, such as large areas dedicated to this, but in reality, many of these plants were interspersed with crops.

The number of different edible crops grown per year varied from 14 at US2 to 129 at US3. Our definition of 'crop' was aggregated by varieties, and this analysis was limited by the level of detail that farmers used to record crop types. The most important crops by weight were squash/zucchini (which we grouped for all farms since some farms grouped them in their records), tomato, lettuce, cucumber, and collard greens. These were also the most frequently grown crops, appearing at 6, 6, 5, 6, and 4 farms, respectively. The breakdown of crops harvested per farm is shown in Figure 4.1. Crops with the largest harvest that contributed up to 75% of the harvest to each farm are shown, and the remaining crops were aggregated into a category named "Remaining". Note that US3 was not included in the figure because harvest weights per crop were not available, but the list of all crops harvested was available.

4.1.2.8 *Data collection*

Data collection methods varied at each farm, but can generally be characterized as either 1) using data that farms already collected (minority of the data), and 2) working with farmers to define data collection methods to track their practices (majority of the data). Details of these data collection methods, plus secondary data sources, are available in the Appendix. For all farms, data collected represent one year of operation, but different 12-month periods between 2019 and 2021 were used.

4.1.2.9 *Life cycle assessment*

Goals

The goals of this LCA were to 1) evaluate the environmental impacts of diverse types of UA, in different geographies with different motivations; 2) to understand the variation in environmental performance of UA by looking at trends, hotspots, system modeling decisions, and sensitive inventory data; and 3) to test the framework laid out in Chapter 3 in an application to varied case studies.

Scope

The system boundary included everything needed to grow produce, through delivery to the consumer. The included processes are shown in the process diagram in Figure 4.2. The main functional units were:

- 1 kg of produce and
- 1 m^2 of area under food production for one year.

We also provide impacts in the Appendix using a functional unit of:

- 1 m^2 of total farm area for one year and
- 1 m^2 of green area for one year (i.e. area for food production plus ornamental or native plants).

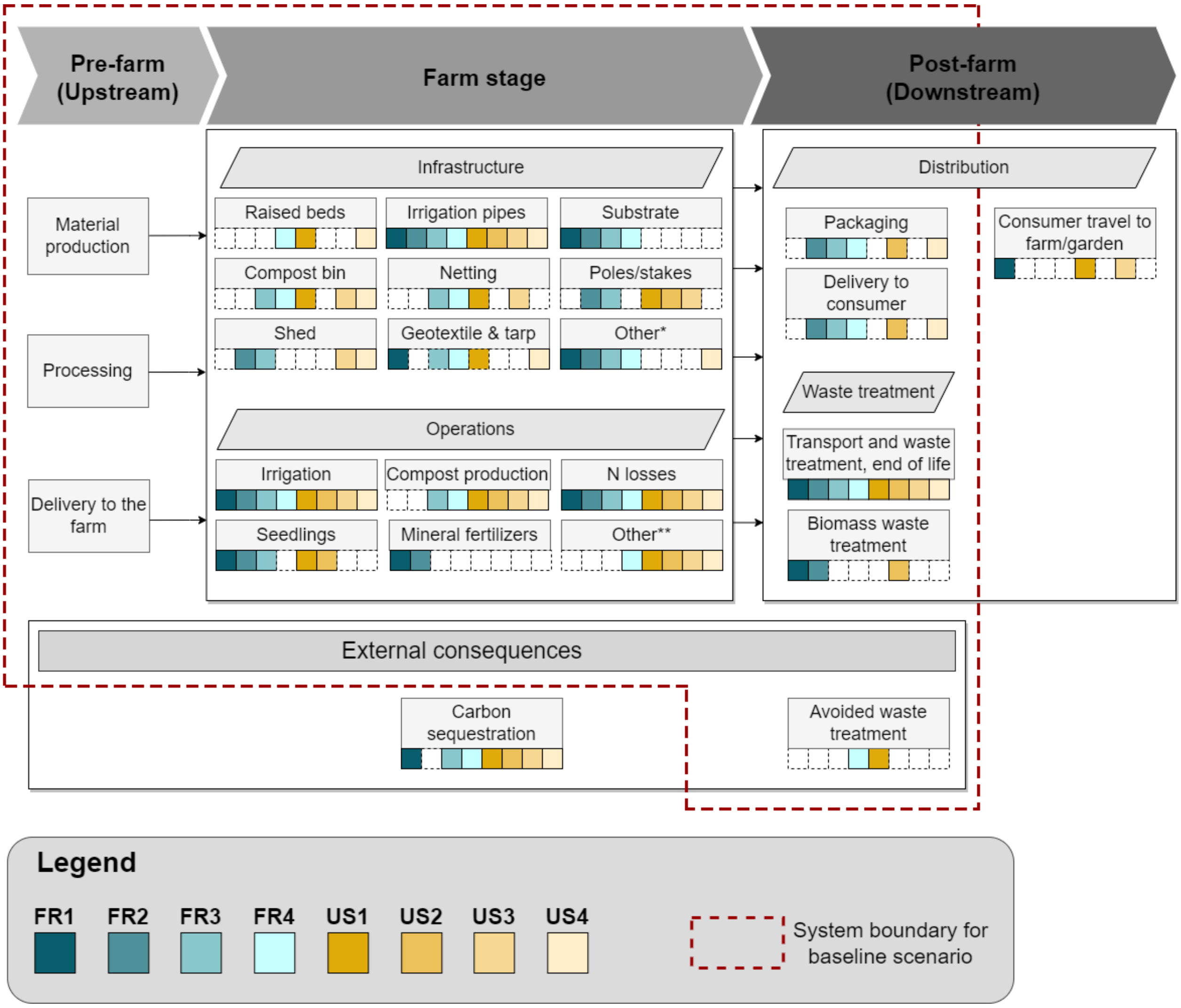

Figure 4.2 The process diagram shows what was included in the system boundaries of the LCA for each farm. Colored squares placed below a process indicate that the process was included for that farm, and a white square indicates that it was not relevant for that farm. Processes outside the red dashed line—carbon sequestration and customer travel to the farm—were included in sensitivity analyses. *Other infrastructure for FR1 was steel frames for vertical growing structures. FR2: hydroponics plastic structure, aeroponics plastic towers, large vat for fertigation mixing, steel tables, and weight distributing tiles. FR3: cables and sand bags. FR4: greenhouse. US4: greenhouse, wood tables. **Other supplies for FR4 were beer brewing residues, mushroom compost, and straw. US1: mushroom compost. US2: fuel for a tractor, crushed oyster shells, and feather meal. US3: wood chips, crushed oyster shells, feather meal, alfalfa meal, and kelp meal. US4: manure, pesticide (Sluggo©), fish emulsion, kelp meal, feather meal.

We used the LCA database Ecoinvent version 3.5, and SimaPro version 9.0.

Life cycle inventory

The processes and inputs at all farms varied, but we categorized them into consistent categories to help interpret the results. The categories and what they included are described in Table 4.2, with details on how they were measured or calculated in the Appendix. Figure 4.2 shows which processes were considered for which farm.

Table 4.2 To help interpret our results, we grouped processes into consistent categories. The name and description of each category is detailed here.

Category name	Description	What it included & details
Substrate	Farms that didn't grow directly in the soil had to create a substrate to grow in.	Material production, processing, and transport to the farm were included. The substrate at FR1 was a lightweight mix of peat moss (25%), coconut fibers (50%) and perlite (25%). FR2 used small amounts of polyurethane, coconut fiber, and peat moss as disposable substrates for the hydroponic and aeroponic systems. FR3 used compost only. FR4 used a mix of 30% wood chips, 30% mushroom compost, and 40% compost. The impacts of substrate were allocated to one year, over an assumed 30-year lifetime.
Infrastructure	Large and semi-permanent or permanent objects used for growing food, such as raised beds, netting, or irrigation pipes.	Material production, processing, transport to the farm, and end of life waste treatment were included. Materials included mostly steel, wood, or various plastics. Impacts were allocated to one year using standard lifetimes for materials and objects, which are available in the Appendix. Infrastructure for activities other than growing food were excluded. For example, tables for hardening off seedlings were included, but picnic tables for leisure were excluded. Infrastructure that was already in place was also excluded, such as sheds in some cases. Specific pieces of infrastructure for each farm are presented in Figure 4.2.
Delivery of inputs	Delivery of annual or regular inputs to the farm.	Delivery of compost, fertilizers, wood chips, straw, seedlings, or chicken feathers by truck were included. Actual frequency and distance of trips were provided by farmers.
Compost	Production of compost on the farm or in an industrial composting center.	Direct emissions to the atmosphere of N_2O, CH_4, VOCs, and NH_3 from the microbial decomposition of organic matter were included. Emission factors were taken from Nordahl et al. (2022). We assumed a mix of 1/3 organic fraction of municipal solid waste (OFMSW) and 2/3 yard waste. For compost made and used on the farm, all impacts were given to the farm. For purchased compost, we used economic allocation to distribute impacts between the compost user and the waste generator, resulting in 7.2% of emissions attributed to the compost user (Pepin, 2022). This was the average percent of revenue at composting facilities from compost sales, relative to revenue from waste dumping fees.
Other supplies	Other supplies (mostly crop	Material production and processing were included. Economic allocation was used to distribute impacts

	inputs), such as fertilizers (organic or mineral), pesticides, wood chips, mushroom compost, straw, fuel, or chicken feathers.	between recycled inputs or byproducts and main products. Transport to the farm was included in a different category: Delivery of inputs. Specific supplies used at each farm are presented in Figure 4.2.
Nitrogen loss	Airborne N_2O emissions and leached NO_3 emissions from organic or mineral amendment applications.	Airborne emissions of N_2O from organic or mineral amendments were accounted for using standard emission factors. For organic inputs we used the IPCC Tier 1 emission factor of 0.57% of applied N emitted as N-N_2O (IPCC, 2019), and for mineral fertilizers (only used in hydroponics systems) this was 0.80% (Llorach-Massana et al., 2017b). This also includes N_2O emissions from leached nitrogen, and nitrate leaching to water. Nitrate leaching was accounted for using the IPCC Tier 1 standard emission factor of 24% (IPCC, 2019).
Irrigation	Water used on the farm.	Municipal tap water was used at all farms. European tap water processes were used for all farms due to a lack of local processes in California, and for consistency. This process included water withdrawals, treatment, and distribution through the city, including losses. In some farms some of this water was used for washing produce and drinking water, but we assumed that amount was small compared to the amount for food production.
Seedlings	Purchased seedlings.	All seedlings were purchased for FR1, FR2, FR3, US1, and US2, as opposed to being started onsite. Peat, irrigation, and greenhouse structure were included. We used a process for tomato seedlings from the Ecoinvent database. This included the materials and processing for all inputs, but transport of seedlings to the farms was included in the 'Delivery of inputs' category.
Delivery	Transportation of food products from the farm to the consumer.	Energy consumed from transport to distribute food were included. Type of vehicle, distance, and frequency were modeled specifically for each farm. For FR4 and FR3 delivery was done on foot; and for FR2, US2 and US4, cars were used for delivery. Trips of consumers to the farm, for FR1, US1, and US3, were included in a sensitivity analysis.
Packaging	Packaging used to distribute food.	Packaging included plastic bags, reusable wooden crates, and plastic clam shells. This included the materials and processing for plastic and wood. Three farms—FR1, US1, and US3—used no packaging because customers came to the farm with their own containers.

Avoided biowaste treatment	Avoided waste treatment of organic waste brought from outside the farm to be composted.	This included food scraps from restaurants and farmers' homes, and coffee grains from nearby cafes. Farms were credited with avoided biowaste treatment, using the global mix in Ecoinvent of 53% municipal composting, 44% incineration, and 3% anaerobic digestion. This category was relevant for FR4 and US1.
Waste biomass treatment	For farms that did not make compost at the farm, we assumed their waste biomass was municipally composted.	This included municipal waste collection, and direct emissions to the atmosphere of N_2O, CH_4, VOCs, and NH_3, and leaching of NO_3. This was a multifunctional process—treating waste and producing a soil amendment—so impacts of producing mineral fertilizer with an equivalent nutrient content were subtracted. We assumed that 1 ton of dry compost produced replaced 23 kg of N-fertilizer, 9.5 kg of P-fertilizer, and 9.0 kg of K-fertilizer (F. Corcelli et al., 2019).

Life cycle impact assessment

We used the Product Environmental Footprint (PEF) impact assessment method, version 2.0 (European Commission, 2017), and results using several other impact assessment methods were provided in the Appendix to support future comparisons to other studies. The eight impact categories considered were: climate change (kg CO_2 eq), water scarcity (m^3 deprived), land use (Pt), energy use (MJ), resource use (minerals and metals) (kg Sb eq), marine eutrophication (kg N eq), terrestrial eutrophication (mol N eq), and freshwater ecotoxicity (CTUe).

Sensitivity analyses

We performed sensitivity analyses to test the importance of decisions that were shown to be important in Dorr et al. (2021a), to follow the guidance from Chapter 3 to present some processes as additional results, and test important system modeling decisions. These scenarios included:

- transport of consumers to farm,
- carbon sequestration from compost,
- avoided waste treatment from compost (for farms that didn't collect waste),
- longer lifetime of infrastructure and substrate, and
- all composting impacts given to compost (no economic allocation).

4.1.3 Results and Discussion

4.1.3.1 *What were the impacts?*

Climate change impacts per kilogram of crop ranged from 0.85-3.4 kg CO_2 eq/kg, with a mean and standard deviation of 1.6±0.79 kg CO_2 eq/kg (Figure 4.3a). US4 consistently had the largest impacts using a kilogram-based functional unit, due to their very low yield, followed by FR4 and US3. The lowest impacts were usually seen with FR1, FR2 and FR3. The main exceptions were for water scarcity, where US1 had the third highest impact overall; and for land use, where FR4 had the highest impacts, mostly due to their use of wood for raised beds and straw for mulch (where straw is a byproduct of wheat and gets small impacts from wheat cultivation). For mineral and metal resource use, FR1 had the second largest impacts, and for energy resource use FR2 had relatively large impacts.

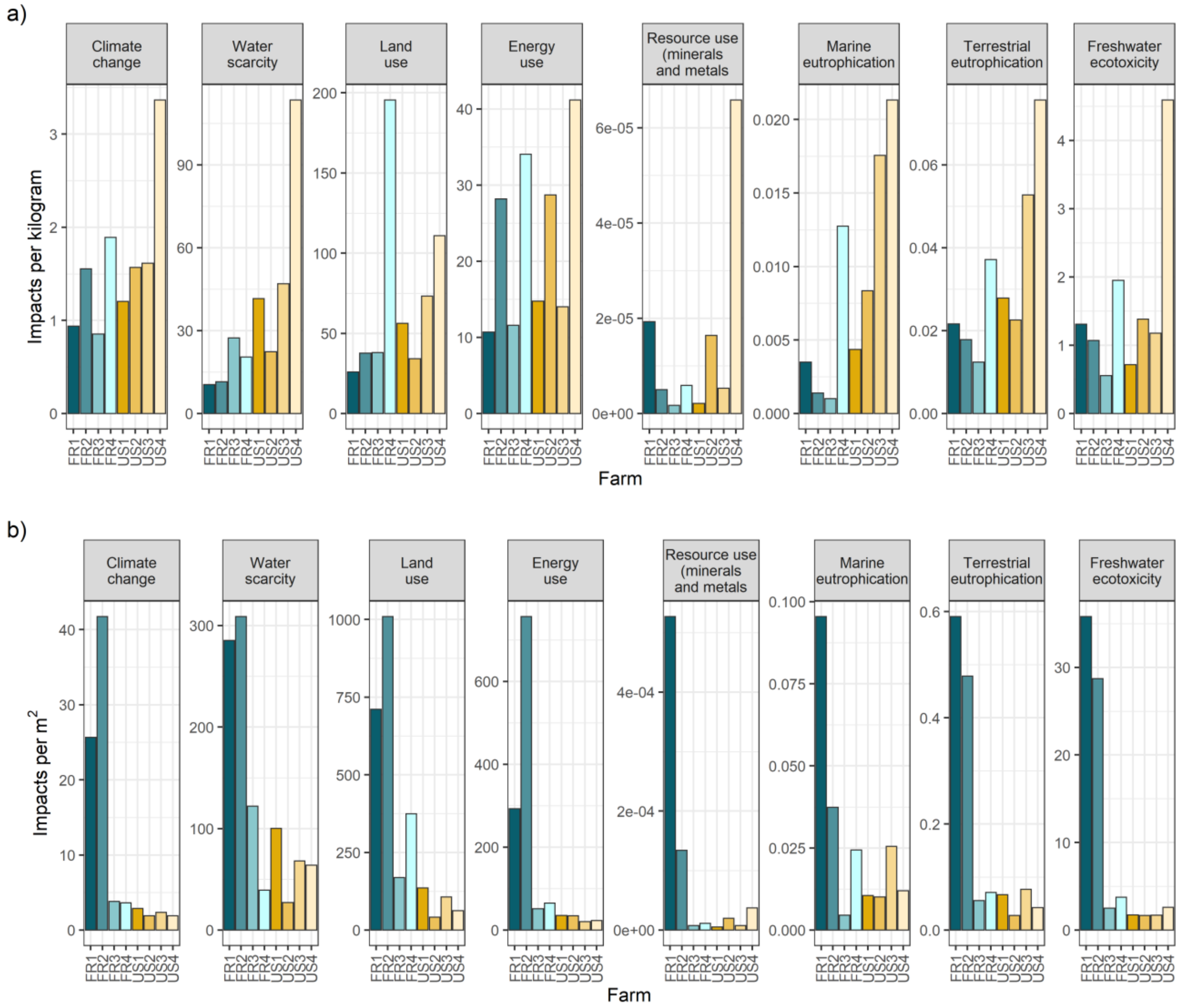

Figure 4.3 Results are shown for eight impact categories with a functional unit of a) kilograms of crop grown and b) m^2 of food growing area occupied per year. The eight impact categories considered were: climate change (kg CO_2 eq), water scarcity (m^3 deprived), land use (Pt), energy use (MJ), resource use (minerals and metals) (kg Sb eq), marine eutrophication (kg N eq), terrestrial eutrophication (mol N eq), and freshwater ecotoxicity (CTUe).

Using an area-based functional unit, FR1 and FR2 had very large impacts, and the rankings were mixed for the other farms (Figure 4.3b). There were different orders of magnitude for climate change impacts per m^2 of food cultivation area between FR2 and FR1 (42 and 26 kg CO_2 eq./m^2, respectively) and the other farms, which had a mean and standard deviation of 2.7±0.84 kg CO_2 eq./m^2. The impacts were so large at FR1 and FR2 because they intensively used space with vertical structures. Considering the more intermediate farms (i.e. all excluding FR1 and FR2), performance was mixed depending on the impact category. For mineral and metal resource use, US4 had the largest impacts due to their shipping container shed. For water scarcity, FR3 and US1 had large impacts because they consumed large amounts of water per m^2.

4.1.3.2 *Trends*

Yield

At the extremes, yield was a highly influential factor determining the relative performance of farms. This was evident for the high-yield farms FR1 and FR2 (with yields 27.4 of and 26.8 kg/m^2, respectively), whose impacts per kilogram were often small, whereas impacts per m^2 were extremely large due to intensive land use. It was also evident for US4 with a very low yield (0.56 kg/m^2, compared to an average of 2.0 kg/m^2 for similar farms, i.e. excluding FR1 and FR2). This led to very high impacts per kg, and moderate impacts per m^2. The other five farms had intermediate yields (1.2-4.5 kg/m^2) and had variable performance, seemingly depending more on their practices and inputs. There may be some tipping points where, to a certain extent, regardless of the practices and inputs, impacts vary depending on the yield. Using a Spearman rank correlation (due to a non-normally distributed, non-linear, and small sample), yield and climate change impacts per kilogram were negatively correlated (ρ = -0.79), and yield and climate change impacts per m^2 were strongly positively correlated (ρ = 0.95).

Variety and type of crops grown

We expected that growing less crops could improve efficiency by focusing on a smaller number of crop needs. For example, when different crops are grown, the amount of water and nutrients provided will meet the needs of the most demanding crop, but be excessive for other crops (Ward et al., 2014). We found no relation between the number of crops grown and the climate change and water scarcity impact per kilogram and per m^2, or yield, according to a Spearman correlation test (p-value > 0.05).

It is difficult to determine the influence of type of crops grown because fruit and vegetable LCAs generally do not distinguish well between different types of crop, on average. In other words, the difference in LCA results between fruits and vegetables is rather small, compared to the differences between fruits/vegetables and other food categories (Lam et al., 2021b; Poore and Nemecek, 2018). The type of crops grown may have influenced US4, which was unique in that their top five crops (by mass harvested) were apples followed by four types of leafy greens (Figure 4.1). Indeed, LCAs for orchards involve more complex modeling that was not possible here with the data available, so we did not treat the apple orchard of the farm separately (Cerutti et al., 2014). Still, apples and other leafy greens tend to have similar climate change impacts per kilogram (Clune et al., 2017). Other farms had variable top crops, but those that consistently emerged as important were tomato, lettuce, squash/zucchini, and cucumber (Figure 4.1).

4.1.3.3 *Process contribution analysis*

Figure 4.5 shows the percent contribution of each process category for all farms. We analyzed the process categories that accounted for the largest share of impacts, and briefly described the categories that were generally not very impactful.

Infrastructure

Infrastructure had the largest average contributions to several impact categories, especially for land use and mineral and metal resource use, where it contributed an average across farms of 43% and 63% of impacts. Its contribution varied across farms, and was especially impactful for FR2, where it accounted for 50% of climate change impacts, 69% of mineral resource use, and 64% of energy resource use. This farm used significant amounts of plastic for the hydroponic structures and the aeroponic towers. US4 also had large infrastructure impacts, mostly due to the shipping container they used as a shed (even though it was severely discounted for the farm, with a long lifespan of 50 years and half of the impacts since it was reused). At US4, infrastructure contributed to 34% of climate change, 84% of land use, 91% of mineral and metal resource use, and 43% of energy use.

We also evaluated the impacts of only the infrastructure for each farm to assess their infrastructure intensity. FR1 and FR2 had total infrastructure impacts that were sometimes 10 times greater than the other 6 farms (for climate change, 1,740 and 6,202 kg CO_2 eq. compared to an average for the others of 245±71 kg CO_2 eq.). The only exception was the land use impact category, where wood had large impacts for US1, US3 and FR4. Because the total impacts may simply reflect the size of the farm, with larger farms requiring more infrastructure and having more impact, we also calculated infrastructure impacts per m^2. The same trend was seen per m^2 of food cultivation area, with FR1 and FR2 usually having much larger impacts. The trend for climate change impacts specifically was even more exaggerated, with FR2 and FR1 having impacts of 21 and 7 kg CO_2 eq./m^2, compared to an average of 0.5±0.2 kg CO_2 eq./m^2 for the other farms. The infrastructure at FR1 and FR2 was extremely impactful, but the high yields that it allowed them to achieve compensated for most of these impacts, and they ultimately had low impacts per kilogram of food grown. There were exceptions for energy use for FR2, and mineral and metal use for FR1, which remained high even per kilogram. The infrastructure at US4 was rather impactful due mostly to their storage container shed, ranking 3rd highest for total impacts from infrastructure; 4th highest in impacts from infrastructure per m^2; and 1st highest per kg.

Irrigation

Water scarcity was completely dominated by water use for irrigation, with a contribution ranging from 90-99% across all farms. Our irrigation category included both tap water and on-farm electricity for pumping, but the majority of impacts came from tap water. The irrigation process category had large impacts on freshwater ecotoxicity, contributing an average of 37%. Here, the potential freshwater ecotoxicity impacts would be from slag for cast iron production for construction of the water supply network (although the local relevance of this is uncertain, since the process in the database models a weighted average of the European water network). Irrigation was the largest contributor to energy use for US1, US3 and US4; and to freshwater ecotoxicity for FR3, US1 and US4. It contributed on average 19% of climate change impacts, but this was as high as 26-31% for US1, US3, US4, and FR3. It contributed 27% to energy resource use on average, and this was 52, 44, and 43% for US1, US3 and US4.

Compost

Compost production for soil amendments was the largest source of terrestrial eutrophication impacts, and the fourth largest source of climate change impacts on average. Among the six farms that used compost amendments, it contributed an average of 57% to terrestrial eutrophication and 17% to climate change impacts. This contribution ranged from 6-32% for climate change impacts. Many parameters with uncertainty were chosen to model compost, and the importance of these was evaluated with sensitivity and uncertainty analyses, in Section 4.1.3.4.

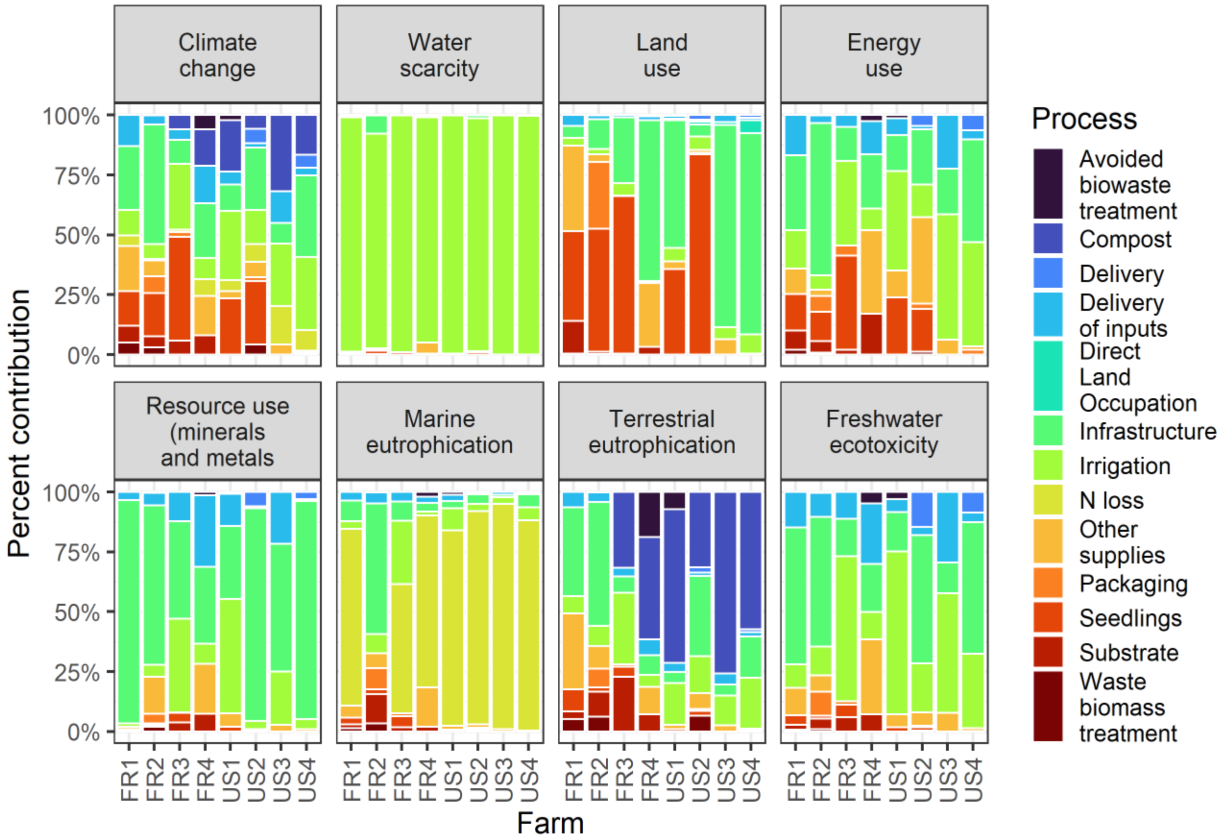

Figure 4.4 The relative contribution of each process category to each impact category is presented. The category "N loss" refers to nitrogen loss. More details on what is included in each category are provided in Table 4.2.

Nitrogen Losses

Nitrogen losses had a major contribution to marine eutrophication due to nitrate leaching, which contributed between 54-94% of impacts (on average 80%). This was excluding FR2, which we assumed had no nitrate leaching due to recirculation of the fertigation water. There is large uncertainty here regarding the actual fate of leached nitrate in urban wastewater systems, and the emission factor of leached nitrate. We used a standard emission factor based on the amount of nitrogen applied, and this is seen as a rough average for rural agriculture systems, but surely has even larger uncertainty for UA growing on different substrates. The marine eutrophication results are very difficult to interpret here, due to the massive uncertainty in the one process that dominates this impact (Pepin, 2022).

N_2O emissions were responsible for 0.5% to 16% of climate change impacts, with an average of 6.4%. The largest contributions were from US3, where emissions from compost and from chicken feathers contributed almost equally. Chicken feathers have high nitrogen contents (about 16% of dry matter), compared to 0.9% for compost assumed here. Indirect N_2O emissions from leaching of nitrogen and subsequent volatilization were responsible for about 30% of these emissions, and direct emissions were responsible for 70%. Similar to nitrate leaching, we used common emission factors here (IPCC Tier 1), so there was large uncertainty in these results, but less consequential than the uncertainty in nitrate leaching due to the relatively small contribution to the impact category.

Seedlings

For the five farms that purchased seedlings, seedling production was important for land use (average 55% contribution), climate change impacts (25%), and energy use (22%). Peat moss was assumed to be the main substrate for the seedlings (according to the Ecoinvent database, and validated as reasonable based on our observations at the farms), and its production was responsible for most of the impacts from seedlings in all of these categories. For the three farms that started seedlings on the farm, we were not able to allocate the inputs used for seedlings, although they surely used small amounts of compost and water. These impacts were accounted for in the final results, but were not isolated to find their contribution to impacts.

Delivery of inputs

The 'Delivery of inputs' category contributed to freshwater ecotoxicity impacts (average of 13%), mineral and metal resource use (11%), energy use (9%) and climate change (8%). These contributions varied largely between farms: for example, from 1-22% for energy use and 2-18% for climate change. Impacts from this process were most important for FR1, FR4, and US3. For FR1, 75% of the delivery of inputs (measured as the product of distance and weight, kilograms × kilometers) was from seedlings, and 25% was from fertilizer delivery. They purchased seedlings from two suppliers 215 and 360 km away, 17 times per year. For US3, most of the delivery amounts (distance times weight) came from compost delivery (78%), and for FR4 this was compost delivery as an amendment (62%) and all material in the original substrate application (28%). Contributions were large here because these farms used (and had delivered) large amounts of compost, but more importantly because compost was delivered from rather far away. The suppliers for these farms were 56-58 km away, compared to other farms where it was an average of 17 km away.

On average, the delivery of inputs was much more impactful than delivery of product, which suggests that there may be a tradeoff in the hyper-local positioning of UA: proximity to the consumer led to low distribution impacts, but this was at the expense of difficulty and distance for delivering agricultural inputs to farms located inside cities.

Other supplies

The 'Other supplies' category was particularly impactful for FR4 and FR1. For FR4, this was partly from the spent mushroom substrate (SMS) purchased from an urban mushroom farm. The inventory for the SMS came from the LCA we did of the mushroom farm (see Dorr et. al (2021b), Chapter 4.2), where we used economic allocation to give SMS a small part of the impacts of the farms' operation (15%). This accounted for 35% of FR4's energy use and 14% of climate change impacts. Straw for mulching was the other main input, and economic allocation was used in the Ecoinvent database to give it a small portion of wheat production impacts (Nemecek and Kägi, 2007). This accounted for 20% of land use impacts at FR4, and 20% of ecotoxicity impacts. At FR1, impacts in this category came from organic fertilizers that were used in their precise fertigation system. Producing these fertilizers accounted for 19% of climate change impacts, and 37% of land use impacts. FR2 also used liquid mineral fertilizers, but used rather small amounts: 0.002 kg N/kg crop, compared to an average of 0.050 kg N/kg crop for all farms (see details in the Appendix). Consequently, fertilizers did not contribute large impacts to FR2. For US2, the supply was gasoline used for a tractor for tilling the soil. It contributed 36% to energy use. It should be noted that this was part of an experiment comparing till and no-till practices, and use of this tractor is not representative of similar urban farms in the area.

Substrate

Substrate contributed an average of 12% of terrestrial eutrophication impacts, 8% of energy use impacts, and 7% of climate change impacts. It contributed the most to impacts at FR4, with 9% of climate change and 12% of terrestrial eutrophication impacts. These impact categories were strongly affected by compost, which was the bulk of the substrate. Substrate impacts from FR1 and FR2 were relatively small, with 5-7% contribution to climate change and 3-10% to terrestrial eutrophication. This was because their substrate was mostly composed of coconut fiber which had no production impacts since it is a waste material. At FR1, substrate accounted for 14% of land use impacts and 8% of energy use, mostly due to peat moss.

Remaining processes

It is also important to note the process categories that were not very impactful here, because the farms studied may have optimized these processes and demonstrate low-impact options, or the processes may be consistently low-impact in UA LCAs and require less attention. These included avoided waste treatment from composting, delivery of the final product, packaging, and waste treatment of nonedible biomass.

Avoided waste treatment from composting was included for US1 and FR4, where organic waste from outside the farm was collected and used for composting. This corresponds to on-farm composting scenario 2, as described in the framework in Chapter 3. At FR4, food scraps were collected from employees who brought them from home, and a nearby restaurant, and collected an estimated 887 kg in one year. US1 collected organic waste from nearby cafes (coffee grounds and food scraps), from gardeners' homes, and leaves from streets that are not usually street-cleaned, totaling 840 kg collected in one year. The impacts of avoided municipal waste treatment of this waste resulted in climate change impact reductions of 6.7% for FR4 and 2.2% for US1.

For delivery, three farms (FR2, US2 and US4) delivered produce very nearby, both by car and on foot, and contributed 0.2, 3.1, and 3.4% to climate change respectively. They contributed 0.2, 4.6, and 6.3% to energy use. Impacts varied by the amount of produce that was delivered using the two modes. Two farms (FR3 and FR4) did all distribution very nearby on foot, and had no impacts from delivery. Our results suggest that proximity to the consumer is indeed a benefit of UA, reducing delivery impacts to zero or nearly zero in many cases (although with a possible tradeoff for delivery of inputs). The remaining farms did not deliver their products, and instead had a pick-your-own model, described in section 4.1.3.4.

FR2 was the only farm with substantial impacts from packaging (wooden crates and plastic clamshells): 28% of land use impacts, 7% of climate change impacts, and 10% of freshwater ecotoxicity impacts. For FR3, plastic crates were reused to deliver produce, and US2 and US4 used thin plastic produce bags. For these farms, packaging accounted for an average of 1.4% of climate change impacts, 2.8% of energy use, and less than 1% of all other impact categories.

Waste biomass treatment was included for three farms (FR1, FR2, and US2) that had no on-farm composting operations, where inedible biomass underwent municipal waste treatment.

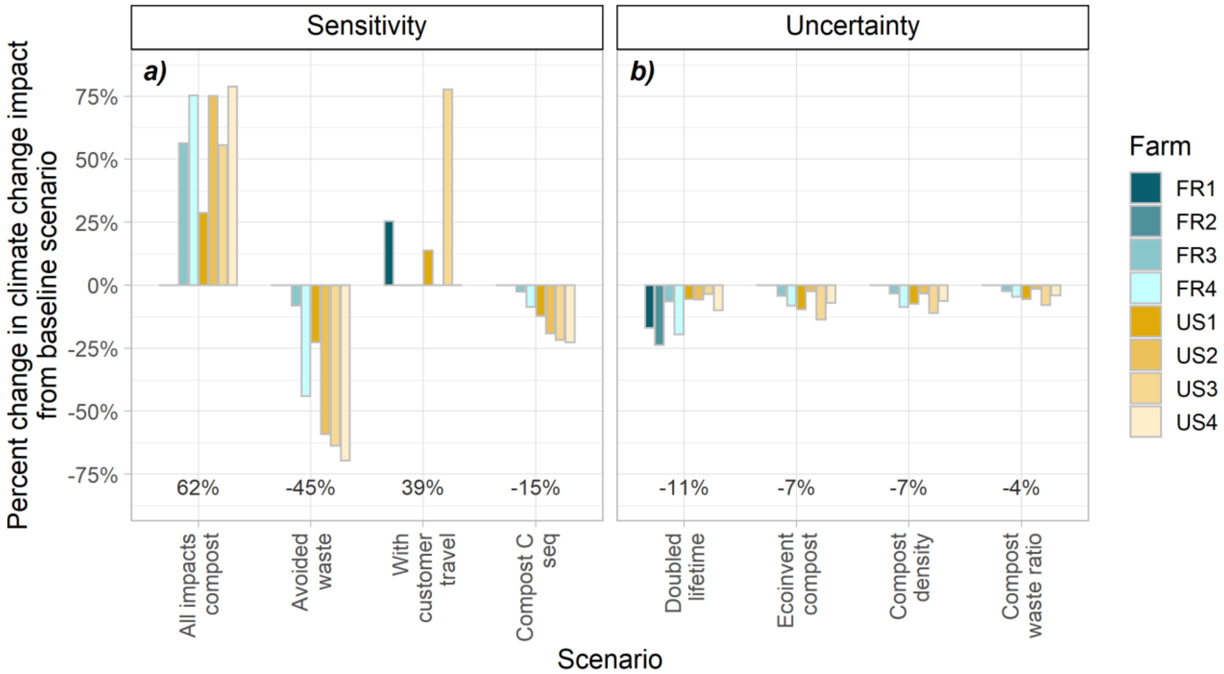

Figure 4.5 a) Sensitivity and b) uncertainty analyses were done to test the effect of different system modeling decisions and parameter values. Bars show the percent change from the baseline scenario's climate change impacts for each farm, and the value shown above the x-axis is the average percent change for that scenario.

This represents scenario 3 described in Chapter 3. This process contributed 3-5% of climate change impacts, 5-7% of terrestrial eutrophication, and an average of 0.3% of other impact categories.

4.1.3.4 *Sensitivity analyses*

Sensitivity analyses were performed to test the effects of our system modeling choices. The scenarios were chosen based on our initial results, and based on recommendations from the guideline presented in Chapter 3. Each modification is described below, and the relative changes from the baseline scenario for each farm are shown in Figure 4.6a for climate change impacts, plus the average relative change.

All impacts from compost

We viewed compost as a co-product of waste management systems, with a share of environmental impacts based on its economic value relative to the other function of treating waste. Specifically, compost was given 7.2% of the impacts of composting. This value came from surveys showing that was the average percent of revenue of composting facilities that came from compost sales, relative to revenue from dumping fees (Pepin, 2022). This is recommended and done elsewhere (Christensen et al., 2018; European Commission, 2010b; Pepin, 2022), but there are also examples where the finished compost product is given all impacts of composting (Adewale et al., 2016; Bartzas et al., 2015). We tested the importance of using economic allocation by modeling a scenario where all impacts from composting were given to the finished product: compost.

This scenario saw the largest changes in climate change impacts, with an average of 62% increase from the baseline scenario among the six farms that used compost. This resulted in climate change impacts of 1.3-6.0 kg CO_2 eq/kg product, and an average of 2.9 kg CO_2 eq/kg product, among the farms that used compost. When all impacts were given to compost, it accounted for an average of 40% of climate change impacts, and 73% of terrestrial eutrophication. Here FR1 emerged as the farm with the lowest climate change impacts (mostly because they didn't use compost), switching with FR3.

Avoided burdens and composting

According to our farm-level scope, the farms should not receive credits for avoided environmental impacts from municipal waste treatment through the use of compost. For purchased compost, these benefits should go to the waste producer, and for on-farm compost, there are no benefits since it was a closed-loop recycling system. Nonetheless, a major supposed environmental benefit of UA is urban symbiosis and the capacity to take up urban organic waste, so we wanted to explore the extent to which this may be important. We modeled avoided biowaste treatment (global average mix of municipal collection, 44% municipal incineration, 53% industrial composting, and 3% anaerobic digestion) for the equivalent amount of waste needed to produce the purchased compost used on the farms. Climate change impacts were reduced by an average of 45% for the six farms that used compost. This ranged from 8% for FR3 which used a relatively small amount of compost, to 70 and 64% for US4 and US3. In this scenario, the hydroponics system FR2 had the largest impacts, and US3 had the smallest. The impacts of US4, which were much larger than all other farms in the baseline scenario at 3.4 kg CO_2 eq/kg, were brought in line with other farms at 1.0 kg CO_2 eq/kg. Our results suggest that this function of UA could be important, but to properly address this topic, a city scale or other system boundary should be used. Territorial LCAs would be a useful approach here (Loiseau et al., 2018).

Customer travel to the farm

Three farms—FR1, US1 and US3—didn't distribute products themselves, and rather customers came to the farm to pick them up. This process is usually not included in agricultural LCAs, and has not yet been included in an UA LCA. Customer travel information was very difficult to track, and too unreliable to include in the main results, so we evaluated its inclusion in a sensitivity analysis. We estimated customer travel mode, distance, frequency, and share of the trip dedicated to this purpose based on a limited number of customer surveys and recorded data for each farm. Including our estimates caused large increases to nearly all impact categories: climate change impacts increased for US1, FR1 and US3 by 14, 25 and 78%, respectively. The only change in ranking for climate change impacts was from the farm that gained the largest impacts from this scenario: US3, which became the second most impactful farm with 2.9 kg CO_2 eq/kg. Energy use increased by 16, 31, and 126%, respectively. The reasons for the varied amount of increase was that we assumed different modes of transport for each farm, based on their accessibility by public transport and bike. Other studies including customer travel to the farm have found that this process can be highly impactful (Loiseau et al., 2020; Mundler and Rumpus, 2012).

Carbon sequestration

Including long-term carbon sequestration from compost in agricultural LCAs is controversial (see Chapter 3). In a sensitivity analysis, we evaluated the potential offsets in climate change impacts thanks to carbon sequestration from annual compost amendments. The detailed calculations for carbon sequestration from compost are in the Appendix. This had variable impacts on climate change, depending on how much compost was used at the farm, and how important compost was for climate change impacts. For the four California farms—US1, US2, US3, and US4—climate change impacts were offset by 12-23% thanks to carbon sequestration from compost amendments. For FR3 and FR4 this was 3 and 9%. This scenario resulted in US2 and US3 having lower climate change impacts than the hydroponics farm FR2.

4.1.3.5 *Uncertainty analyses*

Uncertainty analyses were done to test the effect of uncertainty in inventory data and parameters. Similar to sensitivity analyses, these tests were done by modeling alternative scenarios with changes in the inventory data. Relative changes to the baseline scenario for each farm are shown in Figure 4.6b, plus the average relative change.

Infrastructure and substrate lifetime

Infrastructure was found to be consistently impactful for several impact categories, and substrate was important for FR3 and FR4. The impacts of infrastructure and substrate were directly related to their expected lifetimes, because impacts were allocated over the expected lifetime to the one year of study. When we doubled the lifetime of infrastructure and substrate, climate change impacts were reduced 11% on average. FR2 and FR1 had the largest reductions (24 and 17%), which was expected because they had impactful infrastructure. FR3 and FR4 had reductions of 21 and 17% due mostly to increased lifetime of substrate rather than of infrastructure. FR1 became the farm with the lowest impacts, and FR4 was on par with US2 and US3 (all within 0.08 kg CO_2 eq/kg of one another). Land use, which infrastructure contributed greatly to, decreased an average of 21%, and was especially important for FR4, US1, US3 and US4 (reductions between 27-39%) because their infrastructure was composed largely of wood. For nearly all impact categories, the four rooftop farms in France had larger reductions compared to the four farms from California, which was expected because they had larger infrastructure intensities and substrate impacts.

Compost characteristics

We made several assumptions about compost characteristics and used standard parameters for all farms and gardens (values are detailed in the Appendix). Two important parameters were compost density and ratio of waste to compost, since they were directly used to determine the amount of compost used. A density parameter was used because farmers reported compost use in volume, but in the LCA database it was available in mass. A waste ratio parameter was used because most composting LCAs take a waste-treatment perspective, and represent impacts per kilogram of waste treated, and we needed kilograms of compost produced. Changing the compost density from the baseline value of 450 to 350 kg/m^3 reduced the amount of compost used and reduced impacts. For the six farms that used compost, climate change impacts decreased by 3-11%, and marine and terrestrial eutrophication decreased by 7-20%. Changing the waste ratio from 2:1 kg waste to kg compost to 1.5:1 decreased climate change impacts by 1-8% (average of 4%), and decreased terrestrial eutrophication impacts by 8-19%. The only change in ranking of climate change impacts for both scenarios was a switch between US2 and US3, which had very similar impacts.

Compost emissions

Direct emissions from compost have been measured experimentally under many different settings, and a large range of emission factors has been found. In this study, we used the average emission factors summarized in the review by Nordahl et al. (2022) for composting yard and organic municipal waste, but the actual emissions at farms may have been quite

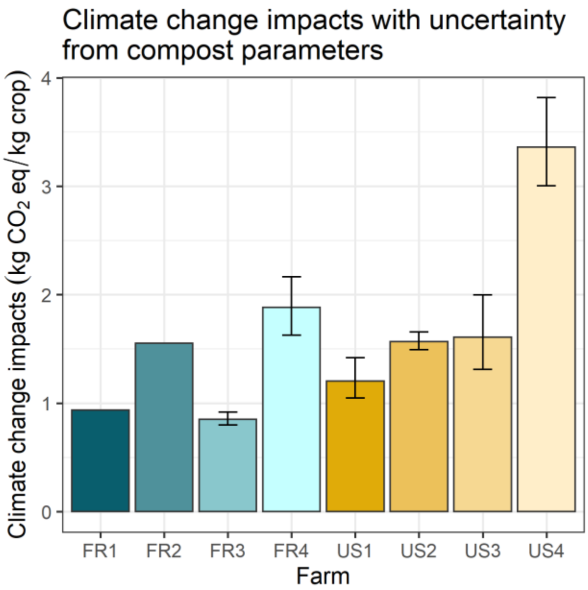

Figure 4.6 We performed Monte Carlo simulations to test the uncertainty of four compost parameters: density, the waste-to-compost ratio, CH_4 emission factors, and N_2O emission factors. The figure shows the climate change impacts of the baseline scenario with error bars representing the 95% confidence interval. Overlapping error bars suggest that farms can be considered to have the same impacts.

different. Even among commonly used sources of emissions factors for LCAs, there is a large variability (see table 3.2 in Chapter 3). We tested our results using a different source of composting emission data: the default composting process in Ecoinvent (Edelmann and Schleiss, 1999; Nemecek and Kägi, 2007). Using the emission factors from Ecoinvent, climate change impacts were reduced by 2-14%, with an average of 7%. Similar to the previous section, the only change in ranking of climate change impacts was a switch between US2 and US3.

Combined compost uncertainty

We performed a Monte Carlo analysis to test the importance of uncertainty from the compost parameters described above. We introduced modest amounts of uncertainty into the LCA model by applying a normal distribution to four parameters. Compost density was 450±50 kg/m^2 and the waste to compost ratio was 2±0.5. Emission factors from composting were given a standard deviation of 25%: N_2O emissions were 0.07±0.018 kg/ton for OFMSW and 0.04±0.01 kg/ton for yard waste; and CH_4 emissions were 0.88±0.22 kg/ton for OFMSW and 2.31±0.58 kg/ton for yard waste. We did Monte Carlo analyses with 1000 runs, and analyzed the climate change impact category. The coefficient of variation (mean divided by standard deviation) was 3-11%. The 95% confidence interval showed a large range of potential impacts for several farms, indicating that some farms can be considered to have no significant differences when we account for this uncertainty (Figure 4.7).

4.1.3.6 *Comparison to other studies*

Compared to other UA LCAs available, this study was particularly valuable because our case studies represented diverse, operational urban farms and gardens, which is often not the case (Chapter 2) (Dorr et al., 2021a). Indeed, LCAs of research systems may be more reliable and straightforward since researchers may have better access to the system and to data, but the real-world constraints and human element of UA are not represented (see Chapter 1).

Because the farms studied here grew unique mixes of many crops, our results cannot be precisely compared with other agricultural LCA results. This is more common for UA, where our review (Chapter 2) (2021a) reported that 17% of UA LCAs similarly studied a mix of crops, and were unable to parse out inputs per crop. Nonetheless, we can compare to LCAs of crops whose harvests were large at the case studies.

Yield

Most of the yields found here were within the range found in other UA LCAs, according to our meta-analysis: yields for open-air UA were usually between 0.5-4.8 kg/m^2, and ours were between 0.6-4.5 kg/m^2, except for FR1 and FR2 with 27 kg/m^2 (yields were the same by coincidence) (Chapter 2) (Dorr et al., 2021a). The average for open-air, soil-based farms in the meta-analysis was 3.3 kg/m^2, and only FR3 had yields greater than that with 4.5 kg/m^2. The yields here were similar to those found in the FEW-meter project (Chapter 1), covering 72 urban farms and gardens, with an average of 1.86 kg/m^2. Overall, we can note that FR1 and FR2 clearly had exceptionally large yields, due to their technical setup. FR3 had rather large yields compared to similar types, possibly thanks to their commercial nature and focus on food production. US4 had a particularly low yield, which could be attributed to several factors: the farm manager was new when data collected started and had experience in landscaping but not food production; the site was in San Francisco which is notoriously cloudy, even compared to the East Bay only 20 km away; crop beds were frequently not replanted soon after a harvest; and other programming activities were prioritized over growing food.

Water

Water use was compared using the amount of water used on the farm, rather than the LCA impact category of water scarcity. This is because there are less studies that use the same impact assessment method that we did (AWARE), and because the 'scarcity' aspect of our results was not very accurate because we lacked appropriate local characterization factors (see the Irrigation section in the Appendix for details), so we used a European geography for all farms. Stone et al. (2021) summarized irrigation amounts for vegetable production in California from the USDA agricultural census, and found an average of 0.89 m^3/m^2. Values from the California farms in this study were mostly larger, and ranged from 0.61-2.0 m^3/m^2, with a mean of 1.3±0.59 m^3/m^2. For water use per kilogram, squash, tomato, and various type of lettuce from rural California had average water use of 0.03±0.01 m^3/kg (Stone et al., 2021). Again, water use in these case California-based studies was higher, ranging from 0.51-2.6 m^3/kg, with an average of 1.3±0.91 m^3. For France, this can be expected around 0.3 m^3/m^2 for rural vegetable cultivation (Grard et al., 2022), which is much lower than what we measured for the French case studies (0.78-6.5 m^3/m^2).

For a comparison to other UA, our meta-analysis (Chapter 2) (Dorr et al., 2021a) highlighted large variation in water use, with a mean and standard deviation of 0.11±0.12 m^3/kg. This was lower than what we found for the case studies. The FEW-meter project also measured much lower average water use (even if there was still substantial variation), with an average of 0.10±0.24 m^3/kg (Chapter 1).

Water use is difficult to compare across geographies due to differences in rainfall, especially here where in 2020, Berkeley, California received 339 mm of rainfall, and Paris received 652 mm in 2019 (NOAA, 2019). It appears that the Paris farms used less water than the California ones: the open air, soil-based farms in Paris had water scarcity impacts of 20 and 27 m^3 deprivation, whereas US1, US2, and US3 had an average of 37 m^3 deprivation, and US4 had an exceptional water use with 113 m^3 deprivation. The FEW-meter project did not find agronomic explanations for variation in water use, and suggest that human factors may be more important (Chapter 1).

Climate change impacts

For climate change impacts, our review (Chapter 2) (Dorr et al., 2021a) found average impacts for intra-urban, soil-based, open-air farms and gardens (ground-based and rooftop combined) was 1.1±1.6 kg CO_2 eq/kg of crop, based on 57 values. Most values (75%) were between 0.07-1.8 CO_2 eq/kg of crop. Among the seven comparable farms here (excluding FR2, which was hydroponics), the average was 1.6±0.57 kg CO_2 eq/kg of crop. Our case studies had comparable values to the averages from the literature, but were on the high end of the range. Only US4, with a climate change impact of 3.4 kg CO_2 eq/kg of crop, had much higher impacts. The coefficient of variation (the standard deviation divided by the mean) was 1.45 for the meta-analysis sample of intra-urban, soil-based, open-air systems, and 0.37 for our case studies. This indicates that there was less variation within our set of results, where farms were still very diverse, than there was between values in the literature.

On an area basis, the average for intra-urban, soil-based, open-air farms and gardens from the literature was 4.2±5.4 kg CO_2 eq/m^2, based on 25 values, with an interquartile range of 0.6-6.9 kg CO_2 eq/m^2 (Dorr et al., 2021a). FR1 had much higher impacts (26 kg CO_2 eq/m^2), but the other six farms had impacts within this range, between 1.9-3.8 kg CO_2 eq/m^2.

FR2 can be compared to open-air hydroponics farms from Chapter 2 (Dorr et al., 2021a): average impacts per kilogram of crop were similar but slightly lower than the average (1.6 kg CO_2 eq/kg at FR2 compared to an average of 2.1±1.7 kg CO_2 eq/kg), and average impacts per area were much larger than the average (42 kg CO_2 eq/m^2 at FR2 compared to an average of 3.9±1.9 kg CO_2 eq/m^2).

Energy use

There were less results available for energy demand: 13 intra-urban, soil-based, open-air farms and gardens had energy demand results summarized in Chapter 2 (Dorr et al., 2021a). The mean was 1.8 kWh/kg, and among our case studies (excluding FR2) values ranged from 3.0-11.4 kWh/kg.

Summary

Overall, our results tended to be larger than what was found elsewhere in the literature, although mostly within a reasonable range. Exceptions where our results were much larger were 1) water use, for most case studies; 2) climate change impacts per kilogram, for US4; 3) climate change impacts per m^2, for FR1 and FR2; and 4) energy demand, for most farms. One reason why our climate change and energy demand results were larger may be that we performed particularly robust, complete LCAs, including many elements that are regularly omitted in UA LCAs. We also used primary data from functioning urban farms, where reality certainly reduces efficiency, compared to ideal research systems often studied in UA LCAs (Dorr et al., 2021a). Real data were collected here, rather than estimates or values from professional rural agriculture, which may suggest that the actual situation of UA is less efficient and more impactful than assumed. Consistent application of LCA and adherence to the guidelines laid out in Chapter 3 also likely led to the lower variability found among our case studies, compared to the variability within similar systems found in the literature.

4.1.3.7 *Understanding variation in our results*

We identified processes and decisions that contributed largely to all impact categories and that drove differences in results. The most important processes were infrastructure, irrigation, and compost, although they contributed differently to various impact categories. We identified five other processes that contributed substantially to impacts: seedlings, other supplies and inputs (like fertilizers), delivery of inputs, and substrate. Nitrogen emissions and leaching were also important, but came with very large uncertainty. The varied performance in the farms studied here can be attributed more to differences in these processes than other less important processes, such as direct land occupation, credits for avoided waste treatment, packaging, delivery to the consumer, and waste treatment of inedible biomass. Yield was important in determining impacts in the more extreme cases (i.e. < 0.6 kg/m^2 and >5 kg/m^2), but the intermediate cases were more complicated. The decision to use economic allocation for the impacts of compost production dramatically affected the climate change results, which would be substantially larger (62% on average) if we had given all impacts to compost. Climate change impacts were sensitive to several high-uncertainty parameters related to compost, which could change the results by 1-14% with small changes in the parameters. Finally, the LCI revealed important variability between years for harvest at farms, which sometimes varied by 50%. For rural agriculture, similar sources of variability have been identified, such as management practices, yield, and year of production (Lam et al., 2021b; Notarnicola et al., 2017). Other sources of variability that were not explored here but have been found to be important for rural agriculture include soil type, climate, and area of production (Lam et al., 2021b; Notarnicola et al., 2017).

The farms studied here varied by some dimensions that were not quantified but are essential for understanding their functioning: the level of social engagement of the farms, and level of professionalization/commercialization. Social indicators were not consistently quantified across farms, but knowing the objectives and general operation of the farms, we applied qualitative levels of social engagement to each farm. Low engagement farms were not usually open to the public or did not hold events that brought in the public, and few people (mostly employees) did the farming. This includes FR1, FR2 and FR3. Medium engagement farms welcomed specific outside groups, usually students, and farming was done mostly by employees and with the help of volunteers, and included US2 and US3. High engagement farms encouraged participation from the public and were farmed roughly equally by both employees and volunteers, and included FR4, US1, and US4. We similarly categorized the farms' level of professionalization, based on the importance of food sales and relative importance of employees vs volunteers. This followed the same pattern as social engagement, with inverse values, where high social engagement corresponded to low level of professionalization, and vice versa. Therefore, we interpreted these characteristics together.

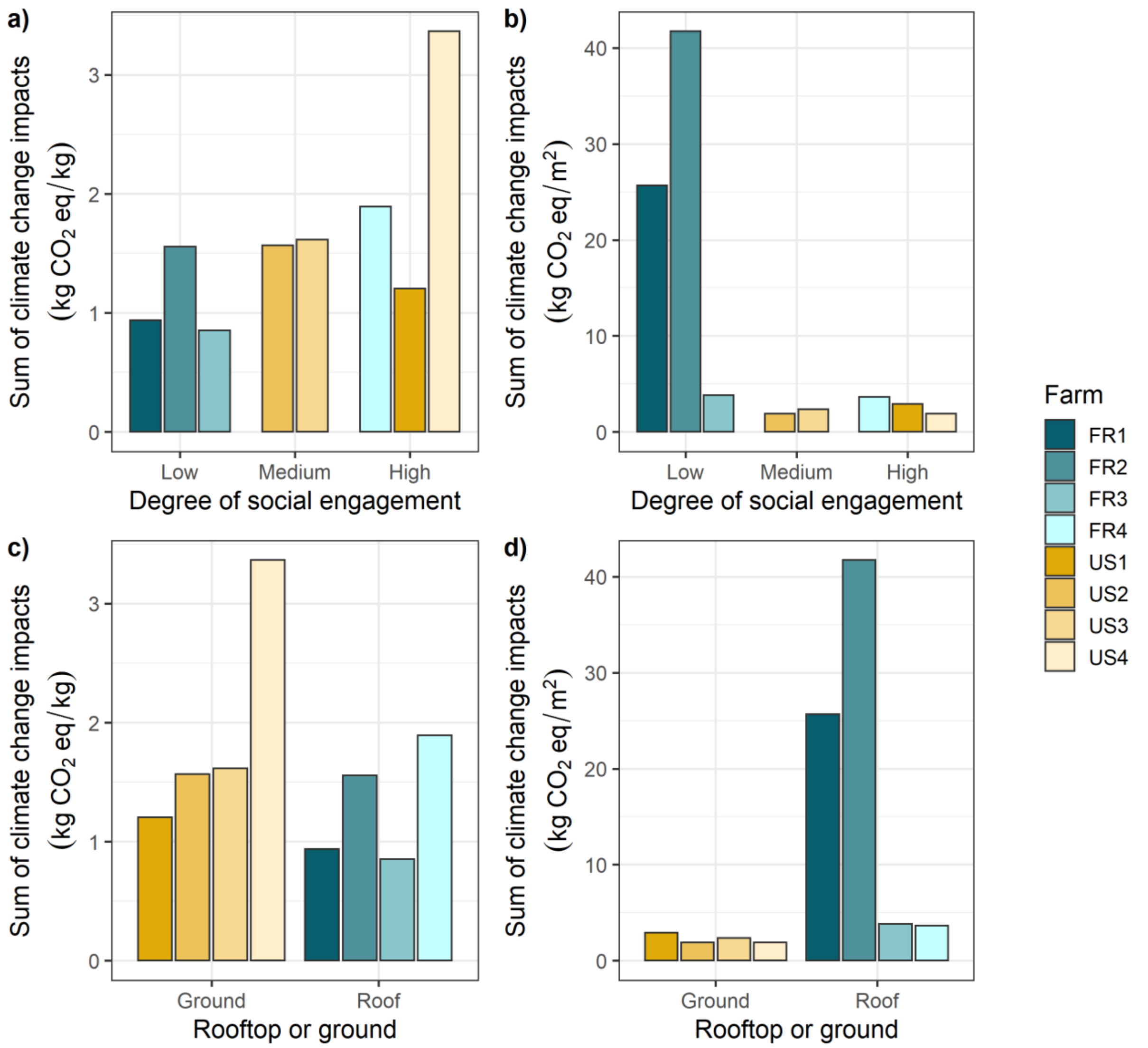

Figure 4.7 Climate change impacts were compared between farms' social engagement level (a and b), and their rooftop or ground placement (c and d). High engagement farms had performed well using an area-based functional unit (b), but had large impacts per kilogram (a). Rooftop farms had larger impacts than ground-based farms considering an area-based functional unit, but this was driven by two of the four farms (FR1 and FR2). Ground-based farms tended to have larger impacts per kilogram.

We found that more professional farms with lower social engagement tended to have lower impacts per kilogram (Figure 4.8a and b), although this effect was compounded with several other factors, especially cultivation setup (i.e. hydroponics/aeroponics and vertical substrate structures). Still, it is clear that the more professional ones used less water and (for FR3) less compost, which were found to be highly impactful processes overall. This may be explained because these types of farms had more centralized systems, which could be run more efficiently by one or several trained farmers with a main responsibility of growing food. US1 appeared to be an outlier, as a farm with high social engagement and very low impacts per kg and m^2 compared to similar farms, partly because of their low reliance on infrastructure. The two school farms, with high social engagement, US4 and FR4, had the first and second largest climate change impacts per kilogram for four of the eight impact categories studied here (but this was not the case for impacts per m^2). We suspect that farms and gardens that had more social engagement had larger impacts per kilogram due to less attention paid to growing food. Here farm managers dedicated large amounts of time to educational programming or managing volunteers, at the expense of growing food. Plus, there may have been trade-offs between efficiency/environmental performance, and farm setup/management to support social engagement, such as lower cropping density, slow crop turnover, or growing in smaller raised beds to improve access to children. Based on our experiences and observations at the farms, we hypothesize that farms with fewer farmers (and therefore a more centralized decision making and crop management system) would be more efficient and have lower impacts per kilogram. Similarly, sites run by farmers and gardeners with more experience may use inputs and space more efficiently. Some indicators that would have been useful to interpret how the level of professionalization and commercialization affected farms' performance include:

- amount of labor required to run the farm (in hours, or full-time equivalent employees),
- revenue from produce sales,
- level of expertise of the farmers and gardeners,
- number of visitors to the farm,
- number of people who participate in farming/gardening,
- hours of volunteer work,
- hours of educational events,
- or percent of farmers' time dedicated to growing food versus other responsibilities.

A major distinction between the physical types of farms studied here was their position on the ground or on rooftops. Farming on rooftops is unique to urban agriculture, so we wanted to assess whether this position may influence environmental performance. We observed higher impacts per kilogram in ground-based farms, and higher impacts per m^2 in rooftop farms (Figure 4.8c and d). However, it was difficult to interpret these results, because this also characterized the California/Paris distinction so there were multiple effects, which seemed more influential than the roof/ground dimension. Other effects included the physical setup (i.e. low-tech vs. medium tech farms), motivation, amount of compost used, and professional orientation. Nevertheless, we can note that farms growing in the ground, in urban soils or creating urban soils on top of an impermeable surface (in the case of US4) applied large amounts of compost to create fertile soils. Indeed, improving the quality of degraded urban soils seemed to be a major concern among these farmers, also reported elsewhere (Edmondson et al., 2014). On the other hand, growing on the roof required FR1, FR2, FR3 and FR4 to create substrate, which contributed moderately to impact categories sensitive to compost for FR4 and FR3. Of the four sites on rooftops, none made structural modifications to the buildings, therefore avoiding large infrastructure burdens. They were constrained to reduce their weight load, which led to the lightweight substrate at FR1 and installation of weight-distributing tiles for heavy fertigation tanks at FR2, but these did not lead to important impacts per kilogram.

A proposed benefit of UA is growing on rooftops and sparing land (Wilhelm and Smith, 2018). Here the ground-based farms had an input of urban land occupation, while the rooftop farms were considered land-free (for the farm itself). We found land use impacts from their direct land occupation were small: an average of 2% across the four ground-based farms (ranging from 0.6-5%). Instead, most land use impacts came from wood for infrastructure, peat for seedlings, and, in the case of FR4, straw. Indeed, if growing on a roof means that wooden raised beds are used (which is often the case, although not always) and the wood comes from forests, then not much land is actually being spared.

4.1.3.8 *Takeaways for different stakeholders*

Here we summarize what our results mean for LCA practitioners and researchers, urban farmers and gardeners, and city leaders.

For LCA practitioners and researchers, our results can help improve UA LCAs. We followed the framework laid out in Chapter 3, and performed thorough analyses of eight diverse farms with a consistent method and system model. We focused on the variation in results, to identify processes that were important and should be included, and processes that come with important uncertainty. Here, we identified compost as a sensitive and potentially important input, which has not been largely studied in UA LCAs so far. Aspects that would be better considered with a city-scale or territorial LCA were identified, such as benefits from composting as an alternative waste treatment, or customer travel to the farm. We can also shed light on some practical difficulties encountered when working with these farms. A major difficulty in data collection was the dynamic nature of UA: farm layouts were frequently changing, new cultivation areas were created, and new farming practices were tested. This made it difficult to capture the average practices for one year. Indeed, for some farms we have data from multiple years, and some factors such as yield varied by 50% between years. There was a high turnover rate in the farmers and managers, who were our main partners for the studies. For half of the farms, the main farmer or point person for the LCA and data collection left during the 1-2 years of collaboration. This raised issues of inconsistency in farming practices, data collection methods, and motivation and willingness to participate in the study. Another difficulty was simply missing records in the data, where food was harvested or supplies were used and not recorded. Farmers were often not used to collecting such information, and this was manual and intensive data collection which required substantial engagement from researchers. Difficulties in data collection with UA have been widely reported in studies aiming to characterize the agricultural practices of UA, let alone perform LCAs (see Chapter 1).

For urban farmers and gardeners, our results suggest how to better manage and design farms to reduce environmental impacts (although we acknowledge that efficiency may not be a main priority for objective for farms and gardens). Overall, our study showed which processes to prioritize, as they are consistently impactful, and which processes may not be worth as much effort. For a simple interpretation, farmers can focus on infrastructure and irrigation, because they were found to be consistently impactful across farms and impact categories. For infrastructure, farmers can prioritize using recycled materials (either through direct reuse or purchasing items made from recycled materials), and using infrastructure for as long as possible. For irrigation, the type of water can be changed to collected rainwater or treated wastewater, which comes with less impacts than municipally-treated tap water. The amount of water may also be reduced by avoiding wasted water through leaks, using timed drip irrigation settings, and avoiding irrigating bare areas that have not been replanted (or, replant on bare areas). Other impactful processes that farmers could optimize are compost and seedling procurement. For compost, farmers can adjust the amount used to ensure they do not use more than is necessary, and purchase compost from facilities that prioritize reducing or capturing fugitive greenhouse gas emissions. Purchased seedlings should be started with a minimum amount of peat.

For city leaders and policy makers, our results may be more meaningful if extrapolated to UA at the city level. Nonetheless, our farm-level resources provide useful findings for UA planning. The performance of different farms can profile which types of UA to promote based on different objectives: if food production is the goal, for example to improve food security of a city, then the more medium-tech farms (such as FR1 and FR2) or professional farms can optimize growing food with lower impacts per kilogram (however we cannot extend this to high-tech farms, since none agreed to participate in the study). If food production is not the main goal, and UA is intended as a means to achieve other goals like education and social benefits, then more low-tech farms are better to minimize impacts per m^2 per year regardless of how much food is grown. The importance of infrastructure in our results gives caution to UA as transient urbanism with temporary installations. If such projects are developed, they should use minimal infrastructure or use recycled material as much as possible. Finally, our results suggest that UA uses substantial amounts of water, although it must be evaluated how important this water use would be compared to what the whole city consumes.

4.1.4 Conclusion

We worked with a diverse set of eight urban farms, collected essential primary data, performed LCAs, and identified which processes and decisions were essential and must be better included in future UA LCAs. We adhered to the guidelines set out in Chapter 3, resulting in an especially comprehensive LCA, and we focused on the sources of variability in the results. Infrastructure and irrigation emerged as consistently impactful processes. Compost, which is not usually focused on in other LCAs and seen as an innocuous input, was important for climate change impacts for five of the eight farms. This highlights the importance of managing composting operations to minimize greenhouse gas emissions. Following this finding, we explored sources of sensitivity and uncertainty for compost, and found that small changes in parameters changed climate change results by up to 14%, and system modeling decisions could increase climate change impacts by 62%. Using two functional units, based on mass of food produced and area cultivated, resulted in different rankings of the farms. Yield was a determining factor of relative impacts between farms in the extreme cases—i.e. with very large or very small yields—but the five farms with more intermediate yields had a mixed performance. Generally, the medium-tech farms (i.e. open-air hydroponics, vertical substrate structures) and the professional farms performed best using the amount of food grown as a functional unit, suggesting that this type of UA may be better for efficiently growing food and alleviating food insecurity. Inversely, they had the largest impacts on an area basis, where the low-tech farms and gardens with more social objectives tended to perform better with an area-based functional unit. This study can help LCA practitioners improve UA LCAs, suggest to urban farmers and gardeners how to reduce the impacts of their cultivation, and help urban planners decide how to support UA in their city to maximize their objectives and minimize environmental impacts. Following the guideline presented in Chapter 3, we obtained results with less variability than what was found in the literature, despite working with very diverse farms, suggesting that application of this framework will help bring consistency to UA LCAs.

4.2 Part 2: Life cycle assessment of a circular, urban mushroom farm

This part presents the life cycle assessment of a mushroom farm in the Paris area, which was published in the Journal of Cleaner Production in March 2021.The only difference between this chapter and the published paper is the numbered references to figures and tables.

Life cycle assessment of a circular, urban mushroom farm

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Abstract

Modern food systems incur many environmental impacts, which can be mitigated by the application of circular economy principles, such as the closing of material and energy loops and the upcycling of waste products. Mushroom farming provides a relevant case in this direction because organic waste can be used for substrate as an input in the cultivation process, which produces valuable outputs such as edible foodstuffs and soil amendment. Few studies evaluate the actual environmental impacts of circular food production systems and assess their efficacy with respect to more linear alternatives. To address this research gap, we quantified the environmental impacts of a circular, urban mushroom farm next to Paris, France. We used life cycle assessment to study the production of 1 kg of fresh oyster mushrooms (*Pleurotus ostreatus*), from the generation of substrate materials through delivery to the distribution center. Our goals were to quantify the environmental impacts of a novel type of food production system, to find the aspects of production that contribute most to these impacts, and to assess the advantages and disadvantages of circular economy for this case study. In terms of climate change impact, the product system emitted 2.99-3.18 kg CO2-eq./kg mushroom, and on-farm energy use was the top contributor to all impact categories except land use. Surprisingly, 31% of the climate change impacts came from transport throughout the supply chain, despite the local nature of the farm. Circular economy actions helped optimize the environmental performance by minimizing impacts from the use of materials, which were mostly upcycled. This suggests that further improvements could be made by reducing energy consumption on the farm or by making the transport schemes more efficient, rather than continuing to focus on the type and source of materials used. This circular, urban farm had similar climate change impacts to classical, more linear systems, but these impacts could be largely reduced by implementing appropriate actions. These were identified and discussed with the farmers, factoring in their feasibility.

Keywords: life cycle assessment; mushroom; circular economy; urban agriculture; industrial ecology; sustainable food systems

4.2.1 Introduction

The current food and agriculture system is considered by many to be environmentally unsustainable due to its substantial emissions, pollution and resource consumption (Campbell et al., 2017). Alternative food systems that ensure the well-being of people and the environment have been put forward (Kloppenburg et al., 1996), which call for improvements in the environmental sustainability compared to the mainstream systems. These can come from extensive and small scale farming, local food production, short supply chains, and circular economy (Forssell and Lankoski, 2015; Kiss et al., 2019).

The latter, circular economy, is particularly relevant in current research, practice, and policy, as evidenced by its major role in the European Green Deal and cities' action plans (European Commission, 2020b; Mairie de Paris, 2017). Circular economy is a principle that comes from the discipline of industrial ecology, which generally aims to design industrial or human-made systems using principles from ecology as a means to attain sustainability (Tóth, 2019). The concept of circular economy emerged from the work of Boulding (1966) as a framework for managing limited resources in a closed system, such as the Earth, and it has gained attention in recent years from academics, policy makers, and the private sector (Merli et al., 2018). Circular economy evokes a departure from linear economies based on "take-produce-consume-discard" models, which assume unlimited resources and waste disposal facilities (Jurgilevich et al., 2016; Merli et al., 2018). Instead, circular economy focuses on closing material, energy and nutrient loops through "reducing, actively reusing, recycling and recovering materials" (Kirchherr et al., 2017). The principles of circular economy are not new, and this paradigm builds upon previous concepts relating to cleaner production, closing loops, and reduce-reuse-recycle (Tóth, 2019). Still, it goes beyond these concepts by considering them in multiple dimensions of sustainability, and by explicitly introducing the notion of full circularity. Scientific studies of circular economy have been done at the macro- (city, region, country), meso- (industrial park) and micro- (consumer, product, company) levels, and are often concerned with the environmental and/or economic sustainability of waste management and the agri-food sectors (Ghisellini et al., 2016; Kirchherr et al., 2017; Merli et al., 2018). Although circular economy can have holistic benefits to environmental, economic and socially sustainable development, we chose here to focus on the environmental dimension (Fassio and Tecco, 2019).

Agriculture has been identified as a relevant topic for implementation of circular economy due to its environmental sustainability issues, large amount of waste production, and important nutrient flows (Fassio and Tecco, 2019). A review of 40 circular practices from case studies in the agro-food sector found that the main circular practices employed relate to optimization, looping, and regeneration (Fassio and Tecco, 2019). Here, optimization focuses on removing waste from production systems by transforming materials regularly considered as waste into valuable inputs to another system without losing value, otherwise known as upcycling. Regeneration refers to a shift to renewable energy and materials, and looping aims to keep materials in closed loops (MacArthur et al., 2015). Within the food system, this can be implemented by utilizing food byproducts and waste to recycle nutrients, avoiding generation of waste altogether, and shifting diets towards foods that can be produced with minimum inputs (Jurgilevich et al., 2016). Collaboration between the food production and waste management sectors is especially important to keep nutrients and organic matter in productive loops rather than discard them as waste through landfilling or incineration.

Mushroom farming is a particularly appropriate activity to demonstrate the potential symbioses of circular economy. Many cultivated fungi naturally cycle organic matter and nutrients by decomposing organic waste and yielding edible mushrooms. The organic waste that mushrooms are grown on is transformed into a nutrient rich soil amendment that is rich in organic carbon, called spent mushroom substrate (SMS) (Grimm and Wösten, 2018; Stamets, 2000). This allows for symbioses in the inputs to the system, whereby mushroom farms can take up waste streams of materials such as straw and manure to give value to the waste and extract their remaining nutrients and organic matter (Sánchez, 2010). For example, Chance et al. (2018) present a mushroom farm that is highly symbiotic with other businesses in an industrial park, through upcycling waste products from beer brewing and coffee roasting. On the output side, SMS, which is essentially composted waste, has many uses as soil amendment, animal feed, biofuel material, wastewater treatment, and packaging material (Grimm and Wösten, 2018; Mohd Hanafi et al., 2018). Oyster mushrooms (Pleurotus spp.) have been shown to successfully grow on waste substrates that do not have other common recycling paths, including grape marc from wineries, waste from olive oil mills, and coffee ground waste recovered after the brewing phase (Koutrotsios et al., 2018; Murthy and Madhava Naidu, 2012). Spent coffee ground (SCG) use is unique because it is an urban waste. Its upcycling by urban and peri-urban mushroom farms would allow for a closed loop system with minimal distance between collaborating actors (waste collection, mushroom production and consumption points), and could place the production near the consumers. Furthermore, an estimated six million tons of SCGs are generated annually worldwide, making up between 16-35% of the food waste from restaurants, cafes and gas stations (Silvennoinen et al., 2015; Tokimoto et al., 2005). Although they can be upcycled by other methods, such as for animal feed, antioxidant extraction, and biofuel, they are typically not valorized and are treated in the regular waste stream (Kovalcik et al., 2018).

Evaluations of circular economy food production are necessary to test the actual environmental advantages of circularity, and to help design optimally sustainable systems. In a review of performance evaluations in this context, Sassanelli et al. (2019) found that life cycle assessment (LCA) was the most commonly used method. LCA is a standardized methodology and tool that models and evaluates systems through their entire life cycle, from extraction of raw materials through disposal (ISO 14040, 2006). Environmental LCA considers the outputs associated with the flows of material and energy in the life cycle of a product, and quantifies the related environmental impacts. Several LCAs of circular food production systems focus on using waste as an input (Dorr et al., 2017; Llorach-Massana et al., 2017a), but to the best of our knowledge, no studies focus on mushroom production. Several studies perform LCAs for current food systems and, based on the outcomes, make recommendations for implementing circular strategies to reduce environmental impacts (Krishnan et al., 2020; Pagotto and Halog, 2016). Comparison between circular and conventional, linear systems points to mixed results, indicating that circular systems should not be considered better by default. For example, Fan et al. (2018) assessed pig farming in a circular agriculture system that also included hay, fish, dragon fruit, mushroom, biogas, and compost production. They found that environmental impacts were higher in the circular system than the traditional system by an average of 43% across 11 impact categories, and that removing some actors from the large network could improve environmental sustainability. Strazza et al. (2015) compared the production of conventional fish feed for aquaculture, made with crops and fish, with a circular option of fish feed derived from food waste, and found that the circular option had lower climate change impacts and energy and water demand by an average of approximately 60%. Also assessing the upcycling of food waste to agriculture, Oldfield et al. (2017) studied the valorization of tomato processing waste for annual preparation of agricultural soils (in a process called biosolarization), and found this circular option to be less environmentally impactful than the business-as-usual system by 20-23%. More LCA case studies in different contexts are needed to evaluate the actual contributions of circular economy agriculture to environmental sustainability.

In parallel, a number LCAs of typical mushroom production have been performed. Gunady et al. (2012) evaluated button mushroom (*Agaricus bisporus*), strawberry, and lettuce production using survey data from farmers in Australia, with a cradle-to-market scope. They found that most climate change impacts in the mushroom systems came from the pre-farm stage, from deliveries of materials for substrate including compost and peat (common substrate materials for button mushroom farming, as opposed to oyster mushrooms which can grow on organic waste). Leiva et al. (2015a) collected data from a button mushroom farm in Spain and performed a cradle-to-farm gate LCA. They found that on-farm energy use was the main driver for all impact categories. Specifically, this was from indoor climate control for most impacts, and from application of compost for climate change impacts. Robinson et al. (2018) performed a cradle-to-farm gate LCA of button mushroom production in the USA. They modeled a typical farm using survey responses from 22 mushroom farmers. They also found that on-farm energy use was the major contributor to several impact categories, and cited use for climate control, trucks, and machinery. Unlike the first two studies mentioned, Robinson et al. (2018) included emissions from the composting process that created substrate to cultivate mushrooms on, and found that it had an important contribution to climate change impacts (23%). The only LCA we found of oyster mushroom production was by Ueawiwatsakul et al. (2014), who collected data from 31 farms in Thailand and used a cradle-to-farm gate scope. The most impactful processes were emissions from burning firewood and fuel to sterilize the substrate, and transport of substrate materials (rice bran and sawdust). The small set of mushroom LCAs show variable CC results, from 2.13-5.0 kg CO2 eq. / kg mushroom, suggesting the need for further research into this type of farming.

To help fill the knowledge gaps in circular agriculture and mushroom farming environmental impacts, we conducted an environmental LCA of a circular, urban oyster mushroom farm in a town neighboring Paris, France. Our goals were first to quantify the environmental impacts of this type of farm and find the most impactful phases of production. Our second goal was to investigate explicitly the circular economy aspects of the farm to understand their positive and negative contributions to environmental impacts. The farm case study grows oyster mushrooms (*Pleurotus ostreatus*) using SCGs collected from Paris as the bulk material for the substrate, in the place of typical substrate materials consisting of agricultural co-products such as straw (Sánchez, 2010). The waste product SMS is sold to local farmers who use it as a substrate amendment, and the mushrooms are delivered to a nearby distribution center in the wholesale market of Rungis and consumed mostly in Paris.

4.2.2 Methods

4.2.2.1 *Case study description*

The mushroom farm is situated on 1000 m2 of land next to Paris in the Yvelines administrative department in France, and sources many materials from and delivers all of its product to the Paris region. Maintaining short supply chains and reusing urban waste to promote a circular economy are important to the farm's mission. This is evidenced first by the upcycling of SCGs. In 2018 alone, the farm used approximately 30 tonnes of SCGs, diverting them from the municipal waste stream of Paris while extracting their remaining organic matter and nutrient contents. The farm's second main contribution to a circular economy comes from waste management, whereby SMS is sold to local farmers who pick it up from the farm and either compost it or directly spread it to agricultural fields as both an organic amendment and a fertilizer. It has even been used by farmers in the urban agriculture network of Paris and shown to be a soil amendment high in organic matter and nutrients (Grard et al., 2015).

The mushroom farm represents a short food supply chain because the major input material (SCGs) is sourced locally (about 35 km. away), the product is sold and consumed locally (about 45 km. away), and there are a reduced number of intermediaries between producer and consumer. The delivery of mushrooms is done daily by an employee who passes near the market every day on his commute home, and so involves frequent deliveries and small volumes. SCGs are delivered to the farm weekly, with the delivery truck returning empty and the frequency of deliveries limited by the amount that they can store, and the risk that large stocks of SCGs sitting on the farm are prone to fungal contamination. Frequent trips with low volumes of material is a regular characteristic of short supply chains, and can be economically and environmentally inefficient (Brunori et al., 2016; Schlich and Fleissner, 2005).

The cultivation of mushrooms follows typical growing practices, requiring approximately 2 months to fruit after being inoculated with mycelium (Sánchez, 2010). The substrate is made up mostly of SCGs, along with wood chips, agricultural lime, mycelium-inoculated rye seeds, and municipal tap water. The substrate materials are mixed, pasteurized using a large autoclave, and inoculated with mycelium, after which the mix is placed in 32 L plastic bags. Next, in the cultivation stage, bags are incubated for about 2 weeks at 70% relative humidity and 17°C and then spend 7 weeks at 93% relative humidity and 16.5°C. During this stage, contamination by competing fungi and bacteria is a major problem, leading to losses of nearly 25% of the bags of substrate prepared. Harvest is done manually throughout these 7 weeks, and occurs several times before the substrate is considered spent. In 2018 a total 8,728 kg of mushrooms were harvested, and during the study period the harvest was 1,253 kg of mushrooms. The mushrooms are packaged in small wooden crates (2 kg per crate) and delivered to the Rungis wholesale food market south of Paris, where they can be sold to local grocery stores and restaurants. The Rungis market is an essential food distribution source for Paris, with 40% of all food consumed in the city passing through Rungis (Mairie de Paris, 2016). The SMS is sold to local farmers who pick it up at the mushroom farm and apply it as a soil amendment.

4.2.2.2 *Life cycle assessment*

4.2.2.2.1 Goals and scope

The main goals of this LCA were to assess several environmental impacts of circular, urban mushroom cultivation, identify the aspects of the system that contribute the largest impacts,

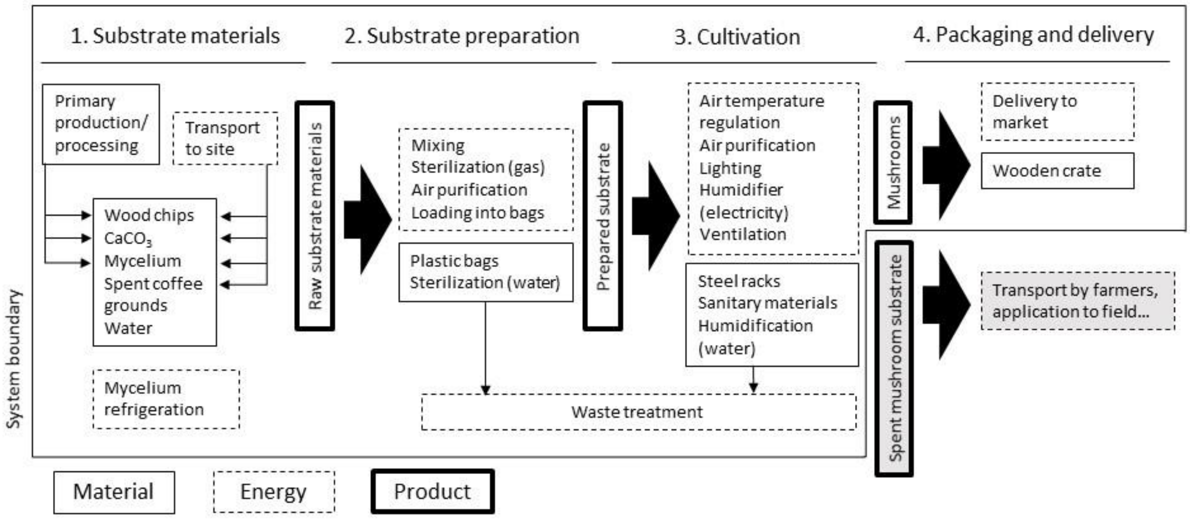

Figure 4.8 The process diagram of production at the mushroom farm shows what was included in the system boundary, and how life cycle stages were delineated.

and evaluate the role of circular practices in the environmental performance. Comparisons to other mushroom LCAs are also presented. The functional unit was 1 kg fresh weight of oyster mushrooms, produced over a 2-month period at the end of 2018. Use of data from a 2-month period was justified because, although there are annual variations in production, they are related to holidays and social factors that affect resource use and food production, rather than climatic conditions. For example, there is lower mushroom production in July and August because of summer holidays, but there is proportionally lower energy and water consumption because of the decision to reduce production. A process-based, attributional LCA was performed, with a cradle-to-market scope. The system boundary is illustrated in Figure 4.9 and includes the extraction of raw materials and energy use used in the foreground and background of mushroom growing, delivery to the distribution center, and the waste treatment of consumed materials. Construction and waste treatment of machinery and infrastructure were excluded due to their assumed longevity and relatively small impacts (Martin and Molin, 2019). Delivery from the distribution center to the final consumer was excluded due to constraints on data collection.

4.2.2.2.2 Life cycle inventory

Background processes were modelled using the Ecoinvent v3.5 database using the recycled content system model (Ecoinvent, 2018). Electricity use was modeled using the French grid. Information about foreground processes was collected from farm records, interviews about farm practices, water and energy bills, and technical specifications documents for machinery and purchased supplies. The life cycle inventory for mycelium production was taken from Leiva et al. (2015b), using Swiss integrated rye production. The life cycle inventory, showing inputs attributed per kilogram of mushroom, is compiled in Table 4.2, and a more detailed inventory with corresponding Ecoinvent process names is included in Table A1 in the Appendix.

Table 4.3 The full life cycle inventory for the production of 1 kg of mushrooms is shown, separated by life cycle stages. The economic allocation between the farm's two products- mushrooms and spent mushroom substrate- has already been applied, giving the mushroom system 84.8% of all material and energy inputs.

Life cycle stage	Input	Material	Value per FU	Unit
Substrate materials	Coffee grounds	Transport, 3.7-7.5 ton lorry (EURO 5)	435.2	kgkm
	Wooden chips	Wood chips, as a byproduct	1.500	kg
		Transport, 3.7-7.5 ton lorry (EURO 5)	145.2	kgkm
	$CaCO_3$	Lime	0.063	kg
		Transport, 3.7-7.5 ton lorry (EURO 5)	0.535	kgkm
	Mycelium	Mycelium inoculated rye seeds	0.358	kg
		Transport, 3.7-7.5 ton lorry (EURO 5)	708.8	kgkm
		Electricity (for refrigeration), French grid	0.012	kWh
	Water	Tap water	1.137	kg
Substrate transformation	Air purification	Electricity, French grid	0.132	kWh
	Conveyor belt	Electricity, French grid	0.079	kWh
	Substrate mixing	Electricity, French grid	0.552	kWh
	Substrate cooling	Electricity, French grid	0.110	kWh
	Sterilization: Gas	Sour gas, global average	5.534	kWh
	Sterilization: Water	Tap water	5.765	kg
	Plastic bags	Polyethylene, low density	0.032	kg
	Air purification	Electricity, French grid	0.188	kWh
Cultivation	Air temperature regulation	Electricity, French grid	4.403	kWh
	Humidifier	Electricity, French grid	0.117	kWh

	LED lighting	Electricity, French grid	1.539	kWh
	Ventilation	Electricity, French grid	0.478	kWh
	Water	Tap water	19.461	kg
	Steel racks	Steel, low-alloyed	0.0082	kg
	Sanitary materials	Polypropylene	0.0007	kg
		Polyethylene, low density	0.0012	kg
		Polyethylene, high density	0.0016	kg
		Synthetic rubber	0.0019	kg
Packaging and delivery	Wood crates	Plywood, for indoor use	0.186	kg
		Transport, 3.7-7.5 ton lorry (EURO 5)	61.801	kgkm
	Delivery	Transport, passenger car, large size, diesel (EURO 5)	0.772	km

To facilitate interpretation of the results, the production system was delineated chronologically into 4 life cycle stages, shown in Figure 4.9. The first stage was "Substrate materials" and included the production and acquisition of materials to compose the substrate on which mushrooms were cultivated, along with electricity from a refrigerator used to store mycelium. Next, the "Substrate preparation" stage involved preparing the substrate materials through mixing, gas-powered pasteurizing, and bagging, along with the plastic bags themselves. The "Cultivation" stage consisted of the inputs used during the 2-month period from inoculation to fruiting and harvest, such as water for cleaning rooms and maintaining humidity, and electricity from LED lights and air heating/cooling. Sanitary materials were counted, including lab coats that were washed and reused 5 times before disposal, and disposable gloves, hair nets and shoe covers. Steel racks that held the hanging bags were also covered here, with an assumed lifetime of 30 years. Finally, the "Packaging and delivery" stage included wooden crates and the transport to deliver products to the distribution center, Rungis, 38 kilometers away.

4.2.2.2.3 Allocation procedures

SCGs used in the substrate were treated using the simple cut-off method (Ekvall and Tillman, 1997) to allocate their impacts to the system that was directly responsible for them, such as the café that used them to make coffee. As a result, the only burdens the mushroom farm is responsible for come from the transport of the grounds from their place of use to the farm site.

The farm produces a co-product along with oyster mushrooms: SMS. Allocation between co-products of a system is a notoriously debated issue in the LCA community, as several options exist but there is no consensus on which approach is best (Finkbeiner et al., 2014). System expansion with avoided burdens is a common and appropriate method, but it can require assumptions that are highly uncertain. According to this method, the system is expanded to include the alternative product that is displaced (or avoided) by the co-product of the system. It is assumed that the system's co-product replaces the alternative product, resulting in negative (or avoided) production of the alternative product and negative environmental impacts (Vadenbo et al., 2017). However, this option would be problematic here because SMS provides many functions and does not clearly replace just one product. It is used as a substitute for composts, mineral fertilizer, or potting soil, and the effect of substituting for each of these products is extremely variable. To avoid making assumptions about such sensitive processes, economic allocation was used to distribute impacts based on the annual revenue from SMS and oyster mushrooms. We chose economic allocation because it appropriately represented the relationship and value partition between mushrooms and SMS. For example, mass allocation would not be appropriate here because the SMS produced has almost six times the mass of the mushrooms produced while carrying only a fraction of the market value of mushrooms. Thus, it appears inappropriate to assign SMS six times more impact than mushrooms. Accordingly, mushroom production at the farm was allocated 85% of the environmental impacts.

4.2.2.2.4 Carbon sequestration from SMS

The mushroom farm transforms a large amount of SCGs (30.3 tonnes in 2018) into SMS (51.3 tonnes fresh weight, 22.0 tonnes dry weight in 2018), which is used as a soil amendment. The SMS at this farm contains 86% organic matter (dry weight), and a significant portion of this is organic carbon (43%, according to Paredes et al. (2009)) that is immobilized in the soil and sequestered, avoiding the emission of CO2 to the atmosphere. Part of this carbon sequestration benefit is attributed to the mushroom production, according to mass allocation between the co-products SMS and mushrooms (85% and 15% by mass, respectively), unlike allocation of impacts of inputs which was done economically. According to measurements of SMS characteristics from the farm and values in the literature (Medina et al., 2012; Paredes et al., 2009), the amount of CO2 eq. per kilogram of mushroom that was sequestered in the soil rather than emitted to the atmosphere was calculated (see details in Appendix 1). The amount of CO2 emissions avoided was entered as a negative emission of CO2 to air in the SimaPro modelling software. Climate change results are presented with and without this sequestered carbon term.

4.2.2.2.5 Life cycle impact assessment

The impacts discussed in this study are climate change (CC), non-renewable energy demand (ED), water depletion (WD), land use (LU), and freshwater eutrophication (FE). These impact categories were chosen because they are important agricultural-related burdens. Additionally, they capture the food-energy-water nexus, which is an increasingly prevalent conceptual framework that highlights the interdependency of these essential resources that have large consumption and are vulnerable in cities (Garcia and You, 2016). The impacts were modeled as midpoint indicators using SimaPro 9.0 software and several impact assessment methods, as described below.

CC, WD, LU and FE were modeled using the Environmental Footprint 2.0 method (European Commission, 2017). The specific methods are the IPCC 2013 100-year model for CC (IPCC, 2013), the EUTREND model for FE (same as the model used in ReCiPe 2008 (Goedkoop et al., 2009)), the soil quality index based on LANCA for LU (Beck et al., 2010), and the AWARE method for WD (Ansorge and Beránková, 2017). Although these methods were selected as the best available, some of them are more accepted than others. WD and LU, for example, were given the lowest recommendation level of 3, which means they are the recommended methods but should be used with caution. The FE model has a recommendation level of 2, defined as needing some improvements, and the CC model received a recommendation level of 1, which is recommended and satisfactory.

ED was modeled using the single-issue characterization method Cumulative Energy Demand V1.11 (MJ), and the sum of the non-renewable fossil, nuclear, and biomass energy demand are used here. It is important to report ED impacts because, although they are generally related to CC impacts, they are not susceptible to variation in local or regional electricity grids, which can have large effects on CC results.

A common issue in the LCA literature is that different impact categories are reported and various impact assessment models are used, rendering results difficult to compare from one study to another. To address this, the results for all Environmental Footprint impact and Cumulative Energy Demand categories are reported in Tables A2 and A3 in the Appendix, although they are not all discussed in this paper. Additionally, results from other common impact assessment methods, ReCiPe 2016 and 2008 (hierarchical, midpoint) (Goedkoop et al., 2009; Huijbregts et al., 2017) and CML (baseline, v4.7) (Guinée et al., 2002) are reported in Tables A4, A5 and A6 in the Appendix for the purpose of comparison to future studies.

4.2.2.2.6 Sensitivity Analyses

Sensitivity analyses are commonly done to evaluate the significance of decisions made regarding the modeling of the system. We performed two sensitivity analyses: first on the electricity grid, substituting electricity mixes for neighboring countries Germany, Spain and Italy – given the unique characteristics of the French mix, with a predominance of nuclear energy. Next, we tested the importance of our decision to use economic allocation for the co-product SMS rather than system expansion with avoided burdens, because allocation is often a sensitive issue in LCA. We compared our results using economic allocation to results from substituting mineral fertilizer for SMS to test how sensitive the results were to this choice. Assumptions and calculations for identifying the quantity of avoided fertilizer from equivalent nutrients in SMS were taken from Robinson et al. (2018).

4.2.2.2.7 Alternative Scenarios

We modeled alternative scenarios to assess how impacts would change if the mushroom production system changed. The first involved a 50% reduction in the frequency of delivery for SCGs, mycelium, and mushrooms, with twice the volume transported each trip. This was to illuminate the potential efficiency issues of transportation in this short food supply chain. It is generally accepted that short food supply chains can suffer from increased environmental and economic impacts from inefficiencies when shipping low volumes of food on the road, which in this case is also coupled with frequent deliveries (Brunori et al., 2016).

The second alternative scenario tested a more typical oyster mushroom substrate: wheat straw. Mushrooms can grow successfully on a wide variety of substrates, and are typically cultivated on agricultural waste or byproducts such as cotton seed hulls, corn cobs, sorghum stalks, or coconut shells. A common and successful material is straw (Sánchez, 2010). From this perspective, the valorization of SCGs as the bulk substrate material at this case study farm is unique to a commercial farm of this size, and is done because of the farmers' commitment to circular economy and the opportunity of being situated nearby a large city with a high concentration of coffee consumption. A comparison was made to production with a more typical substrate composed largely of straw (43% wheat straw, 53% water, 3% mycelium and 1% $CaCO_3$). The life cycle inventory for wheat straw was taken from the Ecoinvent database,

Table 4.4 Life cycle impact assessment results are shown at the level of characterization. Climate change impacts are presented with and without the carbon sequestration contribution from spent mushroom substrate.

Impact category	Value	Unit
Climate change (with C seq.)	2.99	kg CO_2 eq.
Climate change (without C seq.)	3.18	kg CO_2 eq.
Non-renewable energy demand	143	MJ
Land use	169	Pt.
Water scarcity	2.42	m^3 depriv.
Freshwater eutrophication	4.65E-04	kg P eq.

where an economic-based allocation was done to distribute 7-10% of the impacts from wheat grain production (Nemecek and Kägi, 2007). It was assumed that all other on-farm practices and the final yield remained the same. The straw was transported twice the distance as the SCGs (65 km away) because it is not an urban product. It was delivered every 3 weeks, rather than every week for the SCGs, because there is less risk of stocks of straw becoming contaminated.

The third alternative scenario investigated the effect of the overall farm yield by using the maximum monthly value that the farm achieved in 2018. In agricultural LCA studies, where the functional unit is related to food production, results are usually quite sensitive to the yield (Notarnicola et al., 2015). Mushroom farming can have highly variable yields over time due to losses from pests and infection of substrate (Stamets, 2000), and indeed the case study farm incurred losses between 5 and 66% in 2018 (measured in percent of prepared bags of substrate that did not go on to yield mushrooms). According to farmers, the minimum loss rate has been achieved simply through rigorously following the sanitation protocol, including washing hands, wearing lab coats and shoe covers, and keeping doors of the cultivation rooms closed. Average loss rates were used in this LCA study, but since minimum loss rates are achievable with no other changes in production, a scenario with optimal production was modeled using the minimum loss rate recorded in 2018 (5%).

4.2.3 Results

The impacts of production of 1 kilogram of oyster mushrooms are presented in Table 4.3. The percent contribution of each life cycle stage to the overall life-cycle impacts is shown in Figure 4.10. No single life cycle stage dominated all impact categories, but the substrate transformation and cultivation stages were both dominating contributors to several impact categories. The packaging and delivery stage was extremely important in land occupation, and the substrate materials stage generally had modest contributions of 8-27% but was not the major factor in any impact category.

Substrate transformation was the major contributor to CC impacts throughout the life cycle, accounting for 44% of the greenhouse gas (GHG) emissions. In fact, a single process within this stage, gas consumption for pasteurization in the autoclave, accounted for 43% of the CC impacts for the entire life cycle. Substrate materials contributed 24% of the CC impacts over

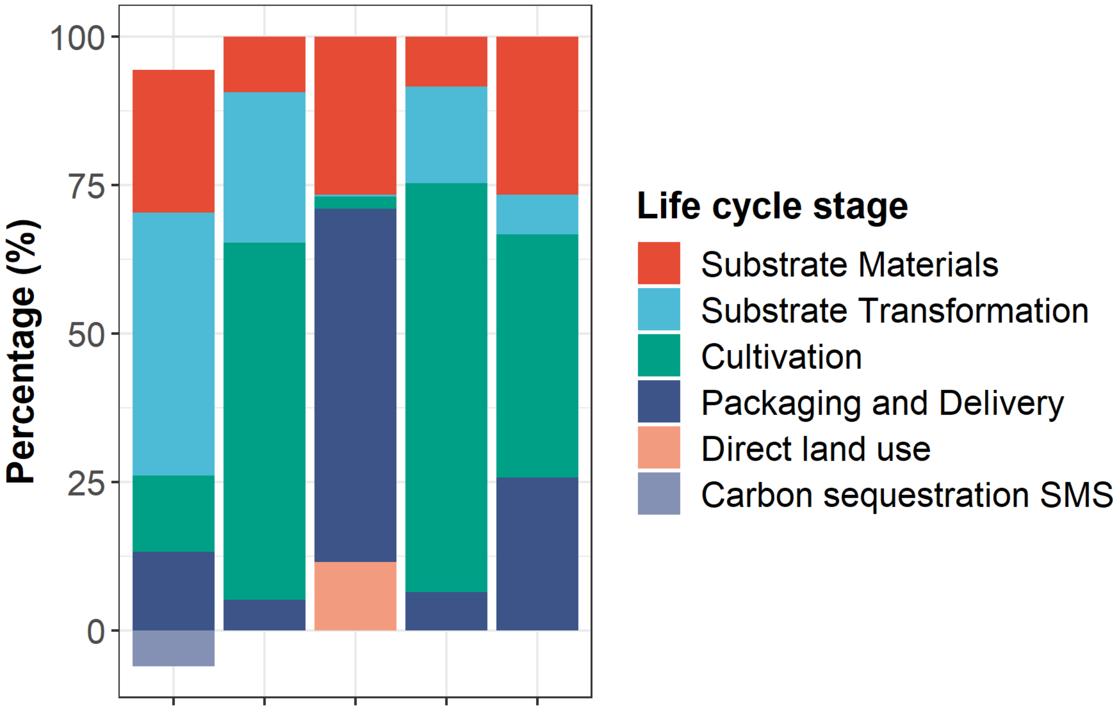

Figure 4.9 The contribution of each life cycle stage to each impact category is shown. The impact categories are climate change (CC), non-renewable energy demand (ED), land use (LU), water depletion (WD), and freshwater eutrophication (FE).

the life cycle, mostly from the frequent delivery of materials. The cultivation stage, which was comprised nearly exclusively of electricity inputs, accounted for 13% of the CC impacts, largely from air temperature regulation. Packaging and delivery of the final product had a modest contribution of 13% to CC impacts, with transport contributing about twice as much as the packaging materials. Finally, carbon sequestration of SMS accounted for 6% of CC impacts.

Contributions to CC were broken down by process type in addition to life cycle stage. The process categories considered were gas, electricity, transport, and materials. Transport included weekly delivery of SCGs and mycelium, infrequent delivery of wood chips and $CaCO_3$, and daily delivery of mushrooms to the market. Material included impacts from producing the materials themselves. Electricity and gas included their use on the farm, and the background processes embedded in the database. The categories of gas, electricity, transport, and material contributed 43%, 14%, 31%, and 12%, respectively (Figure 4.11). Transport from short supply chains, which here were the SCGs and mushroom delivery, contribute 16% of the CC impacts (7 and 9% respectively).

Carbon sequestration from SMS amounted to 0.19 kg CO_2 eq/kg mushroom stored in the soil. This amount was subtracted from the CC impact to give a net CC impact of 2.99 kg CO_2/kg mushroom, which was a 6% abatement. This reduction was rather small because most of the benefits from carbon sequestration were actually allocated to the SMS co-product instead of the mushrooms.

The cultivation stage, with its many electricity inputs, drove the ED with a 60% share. Specifically, air temperature regulation and LED lighting were the largest contributors, with 38% and 13% of the ED over the entire life cycle, respectively. Although gas powered pasteurization drove the CC impacts, which are often closely linked with ED, it only

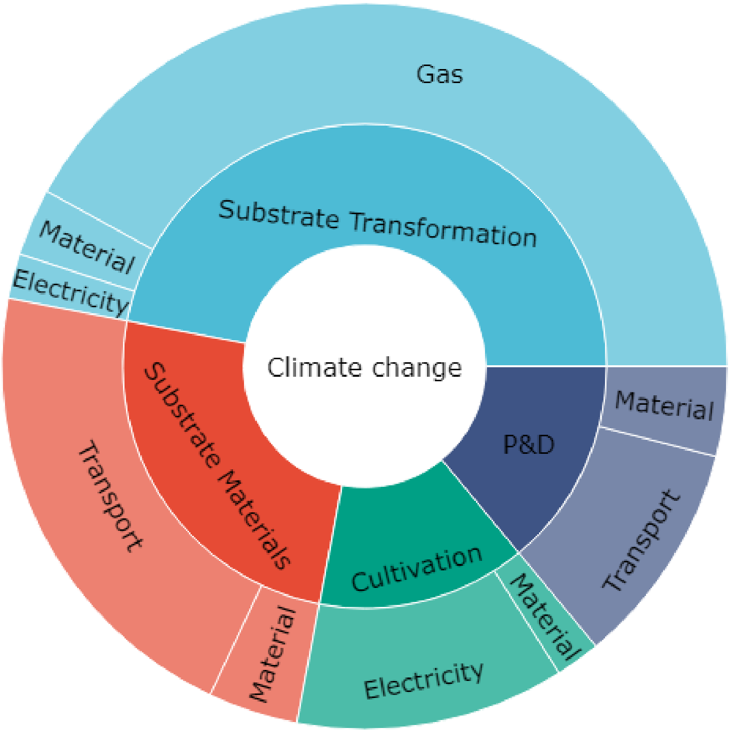

Figure 4.10 The proportion of climate change impacts are broken down by life cycle stage in the inner circle, and by process type in the outer circle. The abbreviation P&D stands for "Packaging and Delivery".

contributed 15% of the ED impacts. This is because the electricity grid in France is largely composed of nuclear energy rather than fossil fuels, so the processes using electricity rather than gas benefitted from low CC impacts (International Energy Agency, 2017).

The direct land occupation of the farm site was small compared to the demands on land in the background system, contributing 12% and 88%, respectively. LU impacts were mostly from wood for wooden crates, used as packaging, which contributed 58% of impacts. The remaining LU impacts came mostly from agricultural production of rye, which contributed 22% of impacts and was used in the production of mycelium for substrate materials.

WD was driven by a variety of different processes with water use occurring in the both foreground and background systems. Most of the contributions came from the cultivation stage (69%), due to water demands from cleaning rooms (where the production rooms are periodically washed down with a hose), humidification of cultivation rooms, and air temperature regulation. The water used for the room cleaning and humidification was tap water used on-site at the farm, while for air temperature regulation the water used was from electricity production in the background system. Most of the water use can be placed in one of 3 categories: electricity, on-site tap water, or embodied water in the wooden crates (Table 4.4).

Impacts to FE were driven by the cultivation stage, mostly from electricity production, with 41% of the total impacts. Other sources of FE came from the transport in the substrate

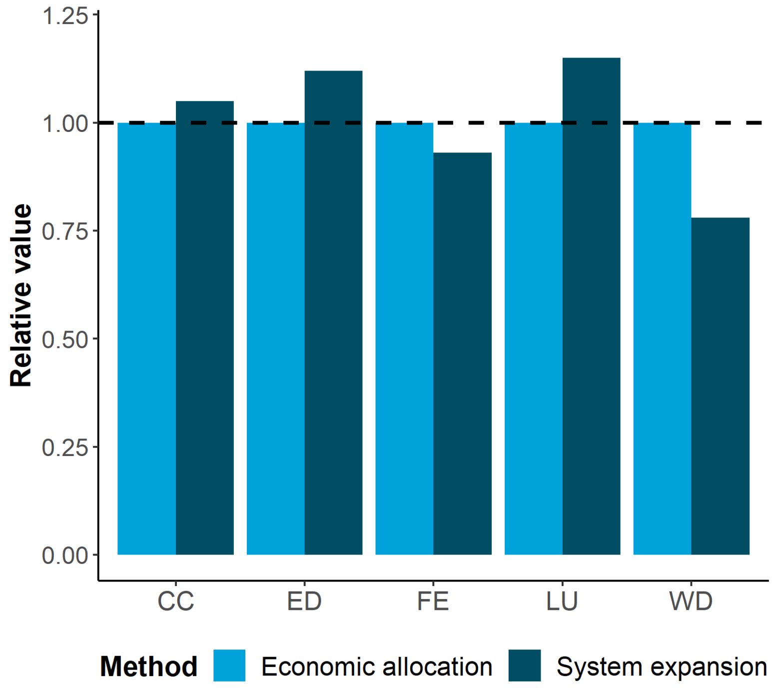

Figure 4.11 Impacts are compared between use of economic allocation (the main method used in this study) and an alternative method, system expansion, to treat the co-product spent mushroom substrate. The impact categories are cliamte change (CC), non-renewable energy demand (ED), land use (LU), water depletion (WD), and freshwater eutrophication (FE). For some impact categories, there is a large difference between allocation methods, and for some there is hardly any difference.

materials and packaging and delivery stages, accounting for 17% and 14% of total FE impacts, respectively.

4.2.3.1 *Sensitivity analyses*

If the same production system were located in and used the electricity mixes of neighboring countries Germany, Italy, or Spain, the CC impacts (with carbon sequestration) would increase to 7.65, 6.00, and 5.29 kg CO_2 eq/ kg mushroom, respectively. However, the ED would decrease by 16-31%, likely due to differing efficiencies of electricity production.

In the second sensitivity analysis, results showed differences of 5-22% in impacts between the two allocation methods, showing mixed responses across impact categories (Figure 4.12). WD was the most sensitive with a 22% difference between allocation methods, whereas CC was the least affected. One method did not have consistently higher or lower impacts than the other, and the choice of allocation system had mixed effects overall.

4.2.3.2 *Alternative scenarios*

In the first alternative scenario we modeled a more efficient transport scheme where deliveries were done less frequently but a larger volume was shipped each time. Despite the farm’s focus on local material sourcing and delivery of mushrooms, there was a substantial impact from short supply chain transport to the total CC impacts (16%). If the weekly deliveries of SCGs and mycelium were cut in half to delivery every 2 weeks, the CC impact (with carbon sequestration) would decrease by 10% to 2.70 kg CO_2 eq. Further reductions of 5% could be made by harvesting and delivering mushrooms every two days, resulting in 2.55 kg CO_2 eq. emitted per kg of mushrooms. These adjustments to the supply chain would result in a net reduction of GHG emissions of 15%.

Next, we modeled a scenario where straw was used instead of SCGs, because it is a more typical substrate material for oyster mushroom production. Production with the straw-based

Table 4.5 There was important water scarcity impacts in the foreground system from tap water use on the farm, and in the background system from electricity generation. Wooden crates, used for packaging, had particularly high embodied water scarcity impacts.

	Substrate Materials	Substrate Transformation	Cultivation	Packaging and delivery	Sum
Industrial water (electricity)	1%	5%	34%	0%	40%
Tap water (on site)	3%	10%	35%	0%	48%
Wooden crates	0%	0%	0%	5%	5%
Other	4%	1%	0%	2%	7%
Sum	8%	16%	69%	7%	100%

substrate had much larger impacts than a SCG-based substrate for FE (33% larger) and LU (784% larger), and slightly larger impacts for WD (6%). The cultivation of straw accounted for a large majority of these impacts, which was expected because they are all closely tied to agricultural production, and straw is a by-product of grain production. CC and ED impacts were lower for the straw-based substrate by 5% and 3%, respectively. CC and ED impacts are not largely changed by this substitution of straw because, like SCGs, straw is a byproduct of another system with little value. Therefore, straw was allocated a minor share of these impacts (7-10%). In both scenarios the CC and ED impacts of materials themselves are small. The delivery logistics of those materials emerge as the more important factor driving impacts, where the straw-based substrate scenario has less frequent deliveries than the baseline SCG scenario.

Finally, we evaluated the impacts of a scenario with realistically increased mushroom yields, using the minimum loss rate recorded on the farm. This linearly reduced all environmental impacts by 43-46%, except for LU, which decreased by 19%. LU responded differently because it is largely affected by wooden crate use for packaging, and the amount of packaging was one of the few inputs that increased with increased in production. For example, the resulting CC impacts with and without carbon sequestration dropped to 1.71 and 1.81 kg CO_2 eq. respectively.

4.2.4 Discussion

4.2.4.1 *Effects of circular economy and short supply chains*

The mushroom farm had low CC impacts from the materials used, accounting for only 12% of the total impact. This suggests that the circular economy model, which was prioritized in the farm design by focusing on upcycling opportunities, was effective at minimizing its impacts. Furthermore, we hypothesized that upcycling of SCGs represented a “more circular” production system than more commonly-used agricultural byproducts such as straw. A comparison to oyster mushroom cultivation with straw showed this was true for some impacts (WD, LU and FE), but other impacts (ED and CC) were not largely affected, because reusing straw (a byproduct) is also a circular system itself. However, the farm-level scope of this LCA did not allow us to model other benefits of using SCGs that would likely be reflected in the CC and ED categories. In particular, the diversion of SCGs away from incineration can generally be considered a net benefit despite a possible energy-generation from incineration (Beylot and Villeneuve, 2013), whereas straw would not be incinerated because it has many applications and its own market. Specifically, municipal waste collection and treatment of the SCGs used per kilogram of mushroom at the farm, using the average French waste treatment mix, would incur an emission of 1.98 kg CO_2 eq /kg mushroom, which is substantial compared to the impacts of using the same amount of SCGs for mushroom production (2.99 kg CO_2 eq /kg mushroom). Additionally, the use of urban-generated waste (SCGs) within urban and peri-urban agriculture can create new links between local businesses and promote innovation. Using this scope of study, it is difficult to evaluate the full advantages of upcycling SCGs.

The circular approach of using SMS as a soil amendment is reflected in the results, in that there were no burdens from waste management and there were some benefits from carbon sequestration. However, the actual impact of avoided waste management of SMS, and the corresponding credits to the farm, are not explicitly shown in our results, according to our modeling decisions. Furthermore, the farm's intentional placement in a peri-urban area nearby the farms that use SMS allows for reduced transport distances, which were not attributed to the mushroom farm given the system boundaries we set.

Regarding the short supply chain aspect of the farm, it appears that the environmental benefits of a reduced distance for transport is offset by frequent trips with small volumes. Average food supply chains have transport processes contributing moderately to CC impacts, with 6-11% through the entire life cycle and specifically 4% from delivery to the final distribution point (Robinson et al., 2018; Weber and Matthews, 2008). Transport at the mushroom farm incurred significant CC impacts, with a 31% share overall, in which 10% came from the final delivery of the product. Although an emphasis is often placed on the delivery of the final product, impacts from transportation of input materials outweighed product deliveries, as has been found in other studies (Martin and Molin, 2019). These contribution calculations only consider the transport in the foreground system, and not transport processes embedded in the database representing the background system, so the actual contribution of transport could be even larger. Our findings support claims that proximity alone is not a sufficient indicator of environmental sustainability, and individual attributes and practices of the system can play a more important role (Edwards-Jones et al., 2008; Kiss et al., 2019; Mundler and Rumpus, 2012).

Overall, processes related to materials from circular economy and transport from short supply chains are not the major sources of impacts across the life cycle. Rather, on-site energy consumption from gas and electricity are extremely impactful. Efforts to improve energy efficiency, or reduce energy use altogether, would likely have more significant benefits to environmental sustainability than making changes to the substrate recipe and changing materials, as the farm currently is focusing on. The most impactful and easiest to implement measures for reducing impacts actually do not require changes in material, transportation or implementing circular economy principles, but adjustments to farmers' behavior to avoid pests and diseases so as to increase the mushroom yield.

4.2.4.2 *Energy source and climate change*

ED at the mushroom farm was relatively high, and was comparable with the ED of greens and herbs in an indoor high-tech hydroponic system (Pennisi et al., 2019). They calculated ED per kilogram in 20 different production systems, and found a range of 53-227 MJ/kg, with an average of 145 MJ/kg, compared to 143 MJ/kg of mushroom found here. Despite this intense ED here, the CC impacts were not proportionally large, compared to other mushroom LCA studies. This is due to the particular electricity grid of France that was used in this study, which is composed of 78% nuclear energy (Ecoinvent, 2018; International Energy Agency, 2017). This allowed for relatively low GHG emissions at the expense of ionizing radiation and other impacts, which were not discussed but are presented in Table A1 in the Appendix. In the case of indoor farming, where large amounts of energy are used, the electricity grid can have a large influence on the resulting CC impacts. In another mushroom farming LCA, Robinson et al. (2018) found important variations in the CC impacts when looking at regions of the USA with different energy grids using more or less coal or renewable energy. Considering LCAs of indoor hydroponic vegetable farming, which similarly use large amounts of energy, Martin and Molin (2019) found approximately 33% increases in CC impacts when using a Nordic electricity mix rather than a Swedish mix in a farm growing basil. In an indoor hydroponic farm growing leafy greens, Romeo et al. (2018) found a decrease in CC impacts of 60% when modeling the difference between the French electricity grid and a wind powered electricity source. This variability highlights the importance of reporting ED in LCAs because this metric is not sensitive to geographic variation in electricity grids.

4.2.4.3 *Comparison to other mushroom LCAs*

It is difficult to directly compare our results to other mushroom LCA studies because most have focused on the common button mushroom (*Agaricus bisporus*), which has different cultivation practices and substrate materials from the oyster mushroom studied here. Additionally, differences in regional and farm-specific practices, background systems, and modelling choices can always lead to differences in results, with unknown importance. Nonetheless, it is useful to cautiously present other mushroom LCA results to position our work.

The only other published oyster mushroom LCA comes from production in Thailand at farms of multiple sizes (Ueawiwatsakul et al., 2014). Our case study is comparable to the small farm size they defined (<20,000 kilograms mushrooms produced per year), and major differences include the substrate, which was composed largely of sawdust in Thailand, and the generation of steam from firewood combustion. Despite these differences, similar CC impacts were calculated, amounting to 3.01 kg CO_2 eq. /kg mushroom (Figure 4.13). However, medium sized farms had larger impacts, of 5.0 kg CO_2 eq. /kg mushroom. They also found large burdens from sterilization of substrate and transport of substrate materials, although due to unique local/regional constraints.

More studies are available for the production of the button mushroom (*Agaricus bisporus*) because it is a more common mushroom. Gunady et al. (2012) assessed button mushroom cultivation in Western Australia and calculated GHG emissions close to ours (at 2.75 kg CO_2 eq./kg mushroom), and found that the largest contribution was from transportation of raw materials, especially the regular transportation of compost from 46 km away. To reduce this impact, they suggested using energy efficient and low GHG fuels, increasing the load factor

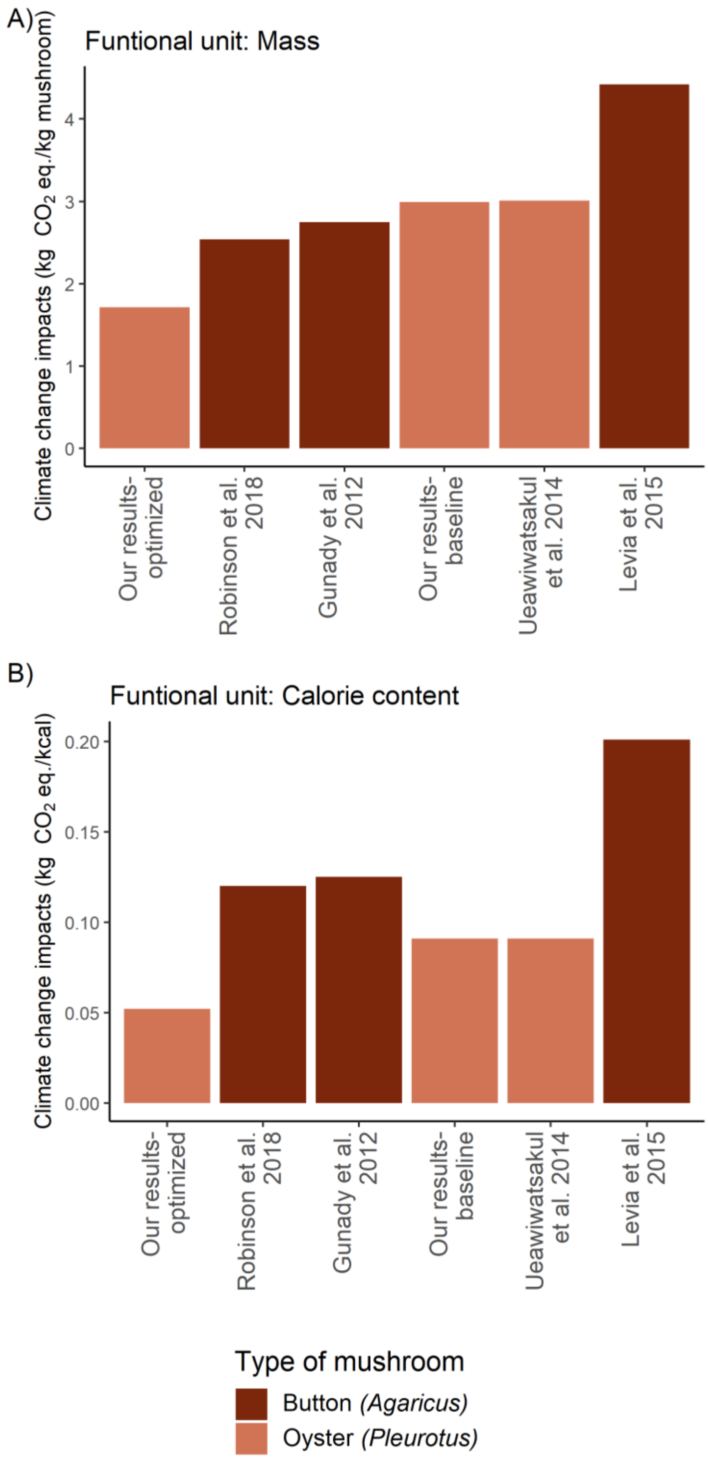

Figure 4.12 Comparing the climate change impacts calculated in this study to the results from other mushroom LCAs showed that the baseline scenario for the circular, urban farm performed similarly to other mushroom farms. However, under the optimized yield scenario, impacts were much smaller at the circular urban farm. When using calorie content as a functional unit instead of mass, oyster mushrooms perform slightly better than button mushrooms.

of trucks to 100%, and avoiding an empty return. They did not mention reducing the frequency of material delivery. In Leiva et al.'s (2015a) LCA of button mushroom production in Spain, CC impacts amounted to 4.42 kg CO_2 eq./kg mushroom, largely due to energy consumption during the growing process and distribution. An LCA of button mushroom production in the USA by Robinson et al. (2018) showed smaller CC impacts between 2.13-2.95 kg CO_2 eq./kg mushroom. Electricity use, fuel consumption and methane from compost emissions made up the majority of the impacts. Total transport emissions only contributed 6-9% of CC impacts, which further contrasts with the high contribution of transport in our study (31%) despite the peri-urban farm using mostly locally sourced materials.

Oyster and button mushrooms have different nutritional and energy contents, with 33 and 22 kcal/kilogram, respectively (U.S. Department of Agriculture and Agricultural Research Service, 2019). Comparing CC results based on energy content, rather than mass, shows oyster mushrooms performing slightly better than button mushrooms (Figure 4.13). This concurrence evidences the robustness of our comparison, and supports the conclusion that CC impacts were within the range of other mushroom farms.

4.2.4.4 *Considerations for LCA modeling*

The boundary of the system excluded delivery to the final consumer, which was a limitation because this can be an impactful stage (Mundler and Rumpus, 2012). Additionally, we used data from the farm for a 2-month period of production, which risks being unrepresentative of the annual production. However, we verified that, although this was one of the most productive periods for the farm during 2018, a proportionally large amount of materials and energy were used as well. Finally, any study on sustainability is limited when it only considers one aspect, where here we focused on environmental sustainability. An inclusion of economic and social aspects would be holistic and ideal, but was outside the scope of this study.

It should be noted that a system modeling choice likely has a large impact here: the decision to treat SCGs, a recycled input, using Ekvall and Tillman's (1997) simple cut off method instead of system expansion and avoided burdens. This choice is necessary because SCGs are a recycled product from the system that created both a beverage in the product's first life cycle, and a mushroom cultivation substrate in its second life cycle. The ISO recommendations for allocation are difficult to apply here (with the following hierarchy: subdivision, system expansion, physical/causal relationships, economic) because the relationship between this primary product and the recycled product is unclear (ISO, 2006b). In this example, if we were to use the system expansion method to include the avoided burden of waste treatment of SCGs, then the impacts of the SCG life cycle must also be attributed. In other words, in order to assign positive impacts (avoided burdens) to SCGs, they must also be assigned their fair share of negative impacts as well. To assign those impacts, an allocation must be done between the coffee grounds for making coffee (product of first life cycle) and the recycled SCGs (product second life cycle). There is no satisfactory way to allocate between these two product life cycles and assign negative impacts, so positive impacts from avoided burdens cannot fairly be assigned, and the cut-off method emerges as the most reasonable solution.

4.2.4.5 *LCA for circular economy*

Several benefits of a circular approach could not be explicitly quantified and highlighted in this study due to our consideration of just the mushroom farm, as opposed to, for example, the cafés producing SCGs and the mushroom farm and the farms applying SMS. One such benefit was the avoided waste treatment of SCGs, which was not included. Additionally, in order to reduce environmental impacts, the farm was established in a peri-urban area to balance distance between urban consumers of fresh mushrooms and peri-urban farmers using SMS. Because the SMS exits the system boundary once the farmers pick it up, this reduced distance was not reflected in the results, although it is a consequence of a choice by the farm. In another LCA of a circular food production system, Strazza et al. (2015) assessed the production of fish feed from food waste on a cruise ship. Taking a similar limited, sub-system only approach, they also did not assign credits for the avoided burden of food waste management when it was upcycled, but acknowledge that the disposal of this organic waste in a landfill would be a significant driver of environmental impacts. Our results suggest that the application of LCA in agricultural circular economy systems is restrictive when applied to an isolated subsystem, such as one farm. Indeed, circular economies are composed of a complex network of actors, and studying only one actor does not capture the beneficial exchanges that may be placed outside of their system boundary and inside the system of another (Zhang et al., 2013). An approach that includes the activities of several actors in a circular economy could be better suited to capture the total advantages of circularity in complex systems (Fan et al., 2018; Oldfield et al., 2017). Therefore, we recommend that when aiming to study circular economy aspects with LCA, a network-level scope should be taken.

4.2.4.6 *Responses from the mushroom farm*

We partnered with a functioning commercial farm and used data from real cultivation practices, rather than a research farm, pilot project, or relying heavily on data from the literature. In addition to the scientific value of this work, we hoped to provide meaningful insight and decision support for the farmers, who were concerned about the environmental sustainability of their practices and looking for feasible paths to improve. An academic-oriented LCA may not naturally generate results that are most interesting to the farmers. For example, because we were interested in the short supply chain aspect of the farm, we modeled an alternative scenario with reduced delivery frequency that reduces CC impacts by 15%. The farmers quickly rejected this strategy because their oyster mushrooms must be delivered daily, as they are the only provider of this specialty product to the market and are constrained by customer demand. SCGs and mycelium could not be delivered in larger quantities because they would not have the space to store them, and because the risk of pathogen contamination would increase. The most feasible improvement, according to the farmers, is the increased yield scenario, where simple sanitary actions by the workers could reduce contamination, attain their highest production rates from 2018, and reduce all impacts by 43-46%. Although they were already aware that they should address the issue of contamination, they said that these results have strongly motivated them and their workers to make it a top priority. One unexpected result was the importance of gas pasteurization to CC, and in response the farmers are exploring ways to mitigate it by contacting the manufacturer of the pasteurization machine to adjust settings, insulating the machine, and installing an electricity-powered machine in a new farm under development. Our experience highlights the importance of partnering with functioning, commercial enterprises and maintaining open dialogues with farmers to consider not only the academic but also the practical outcomes of this type of research.

4.2.5 Conclusion

We conducted an LCA of the production of 1 kg of oyster mushrooms at a circular, urban farm next to Paris. Our goal of quantifying the environmental impacts and identifying the most impactful parts of production yielded valuable results and insight. On-farm energy use emerged as the most important activity for most impact categories, followed by transportation throughout the life cycle. The use of materials had low impacts in most impact categories due to the emphasis put on upcycling in the farm's production design. However, our second goal of investigating the circular economy advantages and disadvantages of the system was met with limited success. This was because our decision to study only the farm as an isolated component of a network of actors excluded several processes that may have large environmental impacts, positive or negative. The tradeoff here was that we were able to study activities at the urban mushroom farm in greater detail, which was valuable because, to the best of our knowledge, an LCA has not been done before on this novel type of food production.

Mushroom farming is indeed a relevant application of circular economy and provides many opportunities for closing material and energy loops. The largest improvements in environmental performance could come from an increased commitment to sanitation practices, which would minimize mushroom losses and maximize yield. The circular approaches adopted at the mushroom farm contributed to environmental sustainability, but on-farm energy use was more important in many impact categories. Compared to more typical mushroom farms studied in other LCAs, this farm had similar CC impacts. However, there is potential for considerably reduced impacts if high mushroom yields can be maintained. Comparing different input materials showed large environmental advantages of using SCGs instead of straw. In some cases of circular food production systems, the most significant enhancements to environmental sustainability may come from efficiency improvements within the system rather than further integrating circular principles.

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Chapter Five: General discussion

5 Chapter 5: General discussion

This chapter aims to discuss the original research presented in previous chapters. First, I synthesize and examine how the main findings responded to the research questions, giving a different perspective to the results isolated in each chapter. This is followed by an outline of the contributions of this dissertation to academia and to stakeholders, thoughts on future research paths and gaps identified and supported by this work, and concludes with my final thoughts and impressions.

5.1 Answering the research questions

5.1.1 Research question #1

My first research question was: how is UA distinct from other production systems in ways that have implications for evaluating them with LCA? This was addressed mostly in Chapter 1, where we evaluated food production and resource use of a diverse set of 72 urban farms and gardens. This was also addressed through work with our nine urban farms and gardens in Chapter 4, but since the sample size was smaller, I drew more upon results from Chapter 1. Simply, the goal here was to get a clearer idea of how UA actually operates. In a second step, I reflected on how these characteristics have implications for LCAs—considering what may be important for UA but not for rural/conventional agriculture and vice versa, and proposing specificities to keep in mind when doing UA LCAs.

The findings are summarized in Table 5.1. Unique characteristics of UA that have implications for LCA are presented. The characteristic is detailed in the second column, based mostly on our observations from Chapter 1 but also from our own LCA case studies from Chapter 4. Observations are from Chapter 1 unless otherwise noted. In the last column, we clarify explicitly how each aspect has implications for doing LCAs.

We were not able to cover all types of UA here, and there were issues with the representativeness of the sample case studies in Chapter 1 and Chapter 4, but this still provided a snapshot of actual practices in open-air UA. This allowed us to make some generalizations about UA later in developing the guidelines, by being able to claim that some things are commonly important or not for UA. This also led us to critically consider some aspects of agriculture LCAs that may not be as significant for UA LCAs. Note that the large-scale characterization in Chapter 1 was limited to on-farm production aspects, and did not consider other processes in the life cycle such as infrastructure, delivery of inputs, and distribution of the produce.

A main limit here was the lack of collaboration with case studies with indoor, controlled-environment growing methods or “vertical farms”. Despite many efforts to contact pilot scale or operational indoor urban farms, we could not find partners willing to participate in the LCA. This was because they did not want to share details about their processes, lacked the time to collect the necessary data, or were not interested in doing such a study. This trend is problematic for a type of agriculture that is promoted largely on its environmental benefits (Benke and Tomkins, 2017; Kalantari et al., 2018). Since we did not have firsthand

Table 5.1 Unique aspects of urban agriculture that have implications for doing life cycle assessments are presented.

Unique aspect of UA	Characterization of the aspect	Relevance for LCA
Variability	There was substantial variation between food production and resource use among sites. Yield ranged from 0.2-6.6 kg/m^2 (average of 1.9), and water use varied from 6.9-646 L/kg (excluding the largest value which was much larger), with an average of 90 L/kg. Compost use was on average 9.6±11.6 L/m^2.	This highlights the difficulty in using averages or expected values for UA production, and foreshadows a similar variability in LCA results for UA.
Compost	Compost was the most frequently used input, but this varied across contexts. All farms/gardens in the UK, the US, and Paris used compost, including all collective gardens. In contrast, only 50% of individual gardens (mostly in Poland and Germany) used compost. This is still much more frequent than rural agriculture. Our LCA case studies used very large quantities of compost (Chapter 4.1).	There are few examples of agricultural LCAs where compost is an input. There is little evidence suggesting how it performs in an LCA. Compost is a particularly tricky input to model in an LCA since it is recycled organic waste.
Synthetic mineral fertilizers and pesticides	These inputs were not very prevalent. Synthetic mineral fertilizers were used in 14% of farms/gardens, with none in the US, the UK and France; and 64% and 26% in allotment plots in Germany and Poland, respectively. Synthetic pesticides were used in 24% of farms/gardens, and were also much more important in allotment plots in Poland, Germany, and France. In contrast, these inputs are common in conventional rural agriculture, and are the focal points of many agriculture LCAs.	Fertilizers emerge as top sources of impacts in agricultural LCAs—especially towards climate change impacts. Pesticides are potentially important for human and ecosystem toxicity impacts. They likely will not be so important for UA LCAs. A major source of potential impacts and uncertainty in agriculture LCAs is the N_2O emissions from fertilizer application. That will also likely be less important for UA LCAs (although N_2O emissions may come from other soil amendments).
Municipal tap water	This was the main source of irrigation water. Groundwater wells were the	Municipal tap water has different environmental impacts from

	second largest source of water, and were used in long-established allotment gardens. Collected rain water was not very important, and no farms/gardens used greywater, despite these sources often being mentioned in UA discourse. In contrast, in rural agriculture irrigation water mostly comes from groundwater wells (Rossi, 2019; USDA, 2018).	groundwater, such as added inputs to treat and transport the water through cities.
On-farm energy use	This was very low or zero. In contrast, in rural agriculture large amounts of energy are used for machinery and for pumping groundwater for irrigation (Barbier et al., 2019).	This input is central to agricultural LCAs but may not require much attention for UA LCAs (for outdoor or non-conditioned systems). For indoor systems, however, energy use has been shown to be a very impactful input (Chapter 2).
Proximity to consumer	Our case studies (Chapter 4.1) showed that many UA projects indeed have close proximity to consumers. Products were often delivered on foot or by bike. When delivery was done by car, distances could be very short. In several cases, customers came to the farm/garden to get products.	Post-farm logistics in UA differ from rural agriculture, where products often travel long distances in supply chains from producer through intermediaries to the consumer. In UA, there may be effectively zero impacts from transport between producer and consumer (when done on foot for example), which raises issues with implied system boundaries. Travel by customers to the site is complex to model in LCAs, and also raises system boundary issues.
Crop diversity	On average 20±16 crops were grown per site in one growing season. In our LCA case studies, this was as high as 129 crops (Chapter 4.1). This represents both spatial and temporal diversity (intercropping and crop rotations).	This results in multiple product outputs from each system which are not necessarily comparable. A functional unit of, for example, 1 kg of tomato cannot be used, as is usually the case in rural agriculture LCAs. Allocation must be done between products, or a different functional unit must be chosen than the typical ‘1 kg of a single crop’.

Unexplained relationships	We could not distinguish links between food production, resource use, and local climate characteristics. This suggests that other factors are important for determining the performance of UA. These factors may be level of experience of farmer/gardener, time spent on food production vs other activities, who is mostly working in the farm/garden (i.e. one trained manager, or children or punctual volunteers), crop choice, or planting density. It could also suggest that inconsistencies disrupt potential trends, such as changes in operations, and turnover of farm managers or gardeners.	This builds on the topic of variability, where not only is there a large range of expected inventory values, but also it is not realistic to extrapolate or assume one value from another. This further underlines the need for data collection from actual UA sites for meaningful LCAs.
Difficult data collection	This work was time-consuming and the participatory approach led to inconsistencies. Eight of the 80 farms/gardens were removed from the study due to data quality issues. Only eight of the 72 sites kept in the study already had data collection methods in place. Researchers regularly checked in with farmers/gardeners, and fixed errors in recorded data. Data collection methods were mostly common, but flexibility and adaptations were necessary to suit each farm/garden. We also experienced great difficulty and time consumption in collecting data from our LCA case studies (Chapter 4).	Data collection is frequently regarded as the most time-consuming step, and barrier, of an LCA. It may be even more difficult for UA due to the diversity of crops and multiple activities/priorities of farmers and gardeners. Plus, issues of uncertainty in the data may be more important here.
Use of space	Large amounts of space were not dedicated to green spaces or food production. Allotment plots in Germany and Poland were mostly used for cultivation (84 and 92%), but less than half of that was for food production (30 and 40%). In contrast, urban farms, community gardens, and allotment plots in the US, the UK, and France had only 30-50% of space in cultivation, but that space was used much more for growing food (85-100%).	This has implications for using land as a functional unit in UA LCAs. Land is the second most commonly used functional unit for agriculture LCAs, after mass of food. LCAs for UA need to specify the scale of land they consider as a functional unit, such as total land, green space, or food production area. Reusing these LCA results for scaling up studies need to account for the type of land considered.

experience with this type of agriculture, our knowledge mainly came from the literature, and the framework lacks a focus on this type of system.

5.1.2 Research question #2

My second research question related to the available evidence and knowledge on UA: what does LCA tell us about the environmental performance of UA? We first aimed to address this through a systematic literature review and meta-analysis—the first one done for LCAs of UA. We summarized the findings in the literature and identified trends. Some main findings were:

- Relatively very few UA LCAs have been done, compared to LCAs of rural agriculture or other sectors. This is an emerging topic of study.
- The available LCAs cover a wide range of UA systems, representing different technical systems (open air or indoor, soil based or hydroponics, ground based or rooftop), different economic orientations (commercial, non-commercial, and research), and many crops.
- A large number of studies evaluated research-oriented systems which seemed unoptimized or innovative. Among these, many covered the same experimental integrated rooftop greenhouse at the Universitat Autònoma de Barcelona in Spain.
- Peri-urban agriculture (PUA) appeared to perform more similarly to rural agriculture and had less variability than UA. However, studies of PUA relied more on data from the literature and from rural agriculture, so this trend may be artificial.
- Water consumption data were not available for most cases. Available results were generally larger than what is found for rural agriculture, and for some cases were extremely large.
- Climate change impacts varied more by technical growing system than by crop.
- Ground based, indoor, hydroponics systems—which can be characterized as vertical farms or plant factories—had the highest yields, but also the highest energy demand and climate change impacts per kilogram of food grown. The increased yields did not overcome the increased resource consumption. Their climate change impacts were especially massive per m^2.
- Ground-based, open-air, soil-based systems—which are the most comparable type to rural conventional agriculture—had similar average yields and climate change impacts to those of similar crops in rural agriculture. However, there was a much larger variability within UA systems.
- There were no obvious differences in the climate change impacts per kilogram between commercial and non-commercial systems.
- For most measures, UA performed similarly to rural agriculture, but was skewed by many cases with much larger impacts/consumption (such as climate change impacts and water consumption).
- UA LCAs frequently cite strong negative correlations between yield and climate change impacts per kilogram. Considering the entirety of the literature, we found no such correlation. Other factors, such as the technical set up or LCA modeling choices, are more important than yield alone.
- On-farm energy use was the most frequent, largest contributor to climate change impacts for ground-based, indoor systems. Greenhouse structure was very important for rooftop, indoor, hydroponics systems (although these studies frequently evaluated

the same greenhouse), and transport of inputs and final product were especially important for ground-based, open-air, soil-based systems.

Next, we addressed this question through our own case studies: first through our LCA of an urban mushroom farm (Chapter 4.2), and later through work with eight urban farms and community gardens (Chapter 4.1). With the mushroom farm in Paris, we evaluated the effect of the local position and low transport distances, and use of spent coffee grounds as a main substrate material instead of the typical material: straw. We found that transport of inputs to the farm and of the product to the market accounted for 31% of the climate change impacts, even though most transport was local (within 40 km). This ended up being impactful due to the frequency of deliveries. Delivery of mushrooms had to be done daily due to market demand, since this farm was the only supplier of this less-common mushroom (oyster mushroom). Delivery of inputs, especially spent coffee grounds, had to be done frequently to avoid storing large amounts, where there would be an increased risk of contamination. Using mostly spent coffee grounds meant the substrate material stage had relatively small impacts, but if we accounted for avoided impacts of incinerating the equivalent amount of spent coffee grounds, climate change impacts would be reduced by 62%. Although most water scarcity impacts came from on-farm water use, as is usually the case in agriculture LCAs (48% here), a large amount also came from upstream electricity production (40%). Ultimately, impacts (or benefits) related to circular economy and local transport were overwhelmed by on-site energy use. By far the largest way to improve the environmental performance was to focus on sanitary measures that would reduce contamination and loss of mushrooms, which is more related to the human and real-world constraints of the system than any technical or material decisions. Climate change impacts per kilogram of mushroom were similar to other mushroom LCAs (although there weren't many available), but if we consider avoided incineration of spent coffee grounds, this system would have much lower impacts.

We came back to this question later in the project through the LCAs of eight urban farms and community gardens (Chapter 4.1). Through our case studies, we demonstrated that infrastructure, irrigation, compost, and peat from potting soil for seedlings were large sources of impacts. Due to the farms' hyper-local positions or setup where customers come to the farm, impacts from delivery were very small. We looked deeper into compost, because it was a main input for these farms and because it is not usually considered in agricultural LCAs. We found that small changes in parameters changed climate change results by up to 14%, and system modeling decisions could change climate change impacts by 62%. The decision to use economic allocation to distribute impacts between the compost product and the waste treatment service was crucial. The farms had very different rankings in impacts based on the two functional units used: kilogram of crop and m^2 food cultivation area. The 'medium-tech' farms (i.e. open-air hydroponics, and vertical substrate-containing structures) had among the lowest impacts per kilogram, but were often the largest per m^2. For these systems, the environmental burdens of added infrastructure and inputs paid off for food production, but due to their large impacts by area they would not be a good model for a system that was not focused on food production. We found that yield seemed to be the determining factor in farm ranking per kilogram for the more extreme cases (very high or low yields), but farms with more intermediate yields had mixed performance that was determined more by their practices or setup. All farms used large amounts of water compared to other UA systems, and much more than rural agriculture—even in Paris, where there was much more rainfall than in the Bay Area. We found that more professional farms with less community engagement tended to have lower impacts per kilogram than non-professional social farms, although for several cases this characteristic was compounded with technical differences (i.e. medium-tech farms).

5.1.3 Research question #3

My last research question related to how we do LCAs of UA. I investigated how the method is adapted to this activity or not, given that UA is a unique activity and methodological reflections are common for LCA to adapt it to a specific sector. This research question was: how should we apply LCA to UA to get the most from it? First indications towards responding to this emerged from the literature review, when identifying what some studies included that others didn't (and should have), the inconsistent system modeling choices, and issues with completeness and transparency. Findings from the literature review that addressed this research question included:

- We found issues regarding the quality of inventory data used, including using data from rural agriculture or averages or estimates. This is problematic since we saw (mostly in Chapter 1) that UA often operates in highly variable, inconsistent, unpredictable ways; and grows a diverse set of crops and has diverse objectives/activities.
- Inconsistency in LCA system modeling led to incomparable systems. Processes were regularly excluded when they seemed important, such as production of substrate and compost, transport to the consumer, direct emissions from amendments and fertilizers, and delivery of inputs. Other 'positive impacts' and avoided environmental burdens were inconsistently included, such as soil carbon sequestration, and avoided fertilizer production or waste treatment associated with composting. Plus, inconsistency in reporting and transparency made the literature difficult to interpret, although this is a universal concern for LCA.
- The small amount of UA LCAs made it difficult to find trends and generalization. This is not a shortcoming per se—it simply reflects the fact that this is a recent and relatively small topic of study. With a larger sample size, results may converge more around reasonable averages with less variation.
- Variability in systems made it difficult to make meaningful, holistic groupings of systems. We attempted to do this using three technical dimensions, and an economic orientation dimension, but still systems within the same group could be very different. Ultimately, there were few replicates of systems that seemed really comparable.

We also reflected on our findings from Chapter 1 (see Table 5.1) and our experience doing LCAs of diverse farms (Chapter 4) in the context of this research question. All stages of the project coalesced to address this question and inform our results here, which were presented as a framework for UA LCAs (Chapter 3). This included practical recommendations, research directions, and discussions raising issues and complexities of doing UA LCAs. The practical recommendations included:

- Ways to address variability (which is especially high) in UA LCAs, such as collecting more data, using distributions, or performing probabilistic simulations such as Monte Carlo simulations. We also recommend the qualitative step of describing UA case studies more holistically (i.e. not only the technical growing setup) to better characterize the system, because UA varies by many more dimensions than only food production.

- For high crop diversity, a basket of crops can be used as a functional unit, or nutritional indexes or land use. Multiple functional units should be used to reflect the diverse functions of UA.
- The multiple objectives of UA are difficult to account for in LCA. Multiple functional units can be used, and holistic descriptions of case studies can give context to results and help in interpretation. Other dimensions should be evaluated, such as social or economic aspects, with life cycle-based methods or other indicators.
- Data collection and availability are limited for UA but are crucial. We recommend specific participatory methods of collecting various types of data, and recommend being transparent and adaptable to ensure successful data collection campaigns with farmers and gardeners.
- For compost, a general discussion of its complexities and considerations was followed by recommendations regarding system modeling decisions and inventory data.
- Creation of substrate was discussed in a first for agricultural LCA methodological reflections. We characterize it as a piece of infrastructure, and present several common substrate materials and their system modeling recommendations. We also address carbon sequestration, annual replenishments, end of life, expected lifetime, and delivery to the farm.
- For transport of the product, we first discuss models of UA where customers come to the farm. This can be very impactful if included, but since it is not usually included and the system boundaries may be inappropriate, we recommend only including it as secondary results. We stress the importance of including transport of produce to the market/consumer when this is done by walking or biking, representing the "last mile" of transport. These processes have negligible impacts, but the system boundary should include them to account for UA's benefit of hyper-proximity to consumers.
- Detailed more in the research directions below, we urge practitioners to consider other non-LCA measures and indicators along with LCA results. These can include social and economic indicators, and other environmental aspects that LCA is not well adapted to include like ecosystem services.

We also identified aspects that should be included or improved for UA LCAs, but are not currently operational, and require more research or methodological development. These include:

- Align UA LCAs with other urban green infrastructure. Most UA LCAs are done in the context of/with comparisons to rural agriculture, focusing on its food production function. UA is equally an alternative urban land use, as much as it is an alternative form of agriculture. Considering it in this context could help city leaders decide which types of urban green infrastructure to implement.
- Use a city or territorial scale to measure the effect of UA in the bigger picture and provide perspective. This can help evaluate if gains in food impacts on a given farm are important at a larger scale, or if resource consumption is important given the vast amount of resources that cities use. This also allows for testing scenarios of scaling-up UA, and projecting the outcomes in food impacts and resource consumption if UA is increased in cities.

- Integrate ecosystem services and LCA for UA. This is useful for improving our understanding of UA, but the greater contribution may actually be towards methodological development for LCA in general. Indeed, there is large interest in integrating ecosystem services and LCA, especially for agriculture, and UA provides a particularly relevant activity through which to develop this integration.
- Explore ways to account for social benefits in LCAs of UA, which is currently a massive omission. Indeed, in many UA projects food is simply a means to deliver social benefits, which are the main objective. Social LCA is a new and developing method to account for the social dimension, but since it focuses on negative impacts embedded through the supply chain and life cycle, it may not be as relevant for UA, where the interest is social benefits at the farm/garden stage. Nevertheless, it would be useful to apply the Social LCA framework to UA.

Finally, we took a step back and asked why we do these assessments. We formulated some questions that UA LCAs may address, based on some common questions in the literature and some new framings:

- Is UA an environmentally positive type of green infrastructure to implement in a city?
- Is UA an environmentally positive way to feed the city?
- Is UA a meaningful way to reduce a city’s GHG emissions?
- If we do UA for social/non-environmental reasons, is it at least not very environmentally harmful?
- Which type of UA should be developed or promoted in a given context (indoor or outdoor, hydroponics or soil-based, commercial or non-profit, professional or volunteer-based…)?
- How can UA be designed or managed to minimize environmental impacts?

Questioning why we do these assessments for UA is indeed a valid concern, where food production or environmental benefits may not be the main objective of farms and gardens. We can imagine community or backyard gardens that grow food mostly as a means to achieve other benefits, and wonder why the environmental performance matters if they are not aiming to improve that, and are operating at a rather small scale. Some of the above questions highlight why LCA would still be useful in such cases.

5.2 Contributions of this dissertation

I hope that the main contribution of this thesis was to shift the direction of research in UA environmental sustainability and impacts. I defined and reframed core questions in this topic, and laid out guidelines for how to do these studies better, so that each study can be more complete, and so the body of literature can provide more reliable and relevant knowledge. My ambitious expectation is that thanks partly to this work, the ‘early days’ and pioneer studies of UA LCAs will transition into a more mature and consistent research topic, as happened with agricultural LCAs. With enough research, of sufficient quality and consistency, results might converge to offer much stronger evidence than what is available now regarding UA environmental performance. Plus, with a stronger foundation of farm-level LCAs, more complex questions can be asked around city-level UA. This can support the development of sustainable cities and a sustainable food system.

A major contribution of this work was the provisioning of primary data from real and diverse urban farms and gardens. Through the FEW-meter project and my own case studies, we covered UA in France; California, USA; New York, USA; the UK; Germany; and Poland. We include farms on the roof and on the ground, in hydroponics and in soil, indoor and open-air, professional and community-based systems (and many systems elsewhere on that spectrum), and cultivation of many different crops (120+) (including mushrooms, which are not largely studied in rural agriculture either). Providing high-quality primary data about the resource inputs and food outputs is extremely valuable for UA. Furthermore, we collected data regarding the amount of water and compost used, which are generally unavailable because they are difficult to track. This kind of data can be used for many other assessments besides LCA.

I provided examples of nine thorough UA LCAs. Given the relatively small amount of UA LCAs, this small contribution is actually quite valuable. These yielded useful findings on their own, and serve as examples for future UA LCAs. We also showed how collaboration with one motivated urban mushroom farm identified feasible paths to largely reduce their climate change impacts.

Finally, this work contributed to the methodological development of environmental assessment of UA, and LCA (separately and together). I highlighted how UA LCAs can be improved, what can be done in parallel to LCA to make assessments of UA more holistic yet practical, what we can learn about UA thanks to LCAs, and what we can learn about LCA through its application to UA. Indeed, due to unique characteristics of UA, it is a challenging topic on which to apply LCA. When such challenges are overcome, highlighted, or broken down and discussed, LCA as a whole can be advanced. Plus, I highlighted where LCA reaches its limits for evaluating UA, and identified aspects where other methods would be more appropriate and feasible.

5.3 Future research

Future research on this topic should aim for **rigorous**, **consistent**, **holistic** studies of **diverse** types of UA. The call for rigorous studies refers to the need for primary data of most operational processes collected from urban farms and gardens. Consistent means more systematically including processes and reporting information so that results can be used elsewhere. Holistic means considering dimensions other than environmental impacts, such as environmental benefits (perhaps through an ecosystem services framework), and social and economic aspects. We highlight the importance of working with diverse types of UA because it is a diverse activity, and research should cover many possible configurations.

There are great opportunities to **learn about LCA through its application to UA**, since UA represents a unique and challenging topic to study. Social LCA here faces the challenge of including social benefits, and the outsized importance of the farm-stage rather than supply chain and life cycle perspective. LCA and ecosystem services could be especially relevant to account for the benefits of UA, at the farm stage. This can include biodiversity and cultural ecosystem services. Researchers develop and test methods to integrate LCA and ecosystem services accounting, and UA is a particularly rich activity to assess this with. Compost offers a great opportunity to reflect on and clarify recycling processes in LCA, since it is a complex process, and the waste-status of composted organic waste is contextual, regarded less as waste as cities implement municipal composting programs.

UA LCAs can be made more nuanced by better **accounting for dynamic temporal changes**. For example, seasonality greatly affects the performance of local food, where local off-season production is less efficient. Seasonality can also affect indoor production, where it becomes too cold to justify growing indoors at a certain outdoor temperature.

Supply chains, logistics, and distribution of UA should be studied, in their description and their environmental performance. Little information is available regarding how UA systems distribute produce, but research from short food supply chains suggests that pitfalls can arise in the frequent distribution of small amounts of produce. Delivery of inputs is even less studied, in UA and other small or alternative farming systems, but results suggest that it can have unique constraints and large impacts. This topic may be considered more from the urban mobility and logistics sector, rather than the agricultural production perspective.

Water use should be focused on more. Very little data exist regarding water consumption in UA, and results from primary data collection suggest it may be larger than what is needed/estimated for growing crops.

Simplified LCA tools for UA should be developed. This was the original goal of this project, but was deemed too ambitious given the recent and varied status of UA LCA research. The data, methodological reflections, and identification of frequently impactful (or not) processes in UA provided in this project can serve as a foundation for development of a tool. These tools have proved useful for rural agriculture, and would be helpful for UA systems that have limited access to/time to collect the full data necessary for an LCA. Plus, UA practitioners may be interested in doing these assessments themselves—evidenced by several that contacted me throughout this project—and a simplified tool is necessary to allow them to do such assessments.

Results should be extrapolated to the **city scale** to evaluate the effect of scaling up UA, and the contributions of UA in its current state. If UA uses twice as much water as rural agriculture to grow similar crops, but this ultimately amounts to a very small amount of water use at the city scale, then research directions may need to shift. Evaluation at the city-scale can also help consequences emerge that are difficult to study at the farm-scale, such as avoided municipal organic waste treatment, or urban transport logistics.

Future research should explore ways to account for **social benefits** of UA and their potential tradeoffs or synergies with the environmental dimension. This can be done ‘within’ LCA using alternative and innovative methods, such as a service-based functional unit (of social services) rather than a product-based functional unit, or using allocation to separate the social and food production aspects. Alternatively, this can be evaluated with social LCA, as discussed above. Finally, indicator-based assessments (non-life cycle) accounting for social aspects can be performed, and results can be evaluated in parallel to LCA results, or integrated with LCA results with multi-criteria analysis or composite indicators.

5.4 Final thoughts

My research was originally framed around and motivated by finding ways to reduce the environmental damages of the food system. As UA is an alternative form of agriculture with proposed environmental benefits, it was selected as the activity to study that may reduce environmental damages of the food system. LCA was selected as the method. I was interested in 1) evaluating the potential of UA to reduce impacts, and 2) the relevance and possibility of LCA to do such evaluations. The second question ended up possibly being the greater contribution of this work, since it had not yet been critically questioned in such detail.

Although my work on the environmental performance of UA was valuable, it was in a way a vehicle to understand and test the work on LCA guidelines, which were the more novel output here.

Regarding my original interest and framing, UA doesn't appear to be a resounding and substantial way to transform the food system towards lower environmental impacts—especially climate change impacts, which my focus was rather limited to due to the lack of and inconsistency in research on other impacts. UA's reduced food miles and often innocuous growing practices are probably not sufficient to provide large gains over rural agriculture. In fact, considering ways to transform the food system, urban vegetable and fruit production does not offer a large margin of improvement to act on: vegetables and fruits can make up relatively small parts of the impacts of food consumption, ranging from 5-20% of climate change impacts from food consumption of cities (González-García et al., 2021; Kim et al., 2020; Supkova et al., 2011). Furthermore, scaled-up scenarios suggest that UA may only replace and potentially reduce impacts from small amounts of fruits and vegetables consumed in cities (Weidner et al., 2019). Indoor systems that suggest benefits towards other types of environmental impacts, such as lower water, nutrient and land use, seem to be overwhelmed by massive energy use and climate change impacts.

Framing this topic as UA's contribution to reducing environmental impacts of the food system, which came from my background and experience in environmental science and agronomy, may not be the most suitable for evaluating the potential of UA. Nonetheless, there are many other facets of UA that justify giving it our attention. For example, a promising source of environmental benefits of UA may be indirect, in the form of social learning by UA participants, who shift towards low-impact behaviors and consumption. Such participants may change their diet by consuming less meat (which has been shown to have the most environmental benefits), or reduce their food waste, or become more sensitive to composting organic waste (Kim, 2017; Puigdueta et al., 2021).

At the same time, we recognize that in many cases, UA probably doesn't have substantial environmental impacts compared to those at the city scale, or of rural agriculture, or of other urban green infrastructure. This prompts my perspective of, if we want to do or promote UA for other reasons, why not? And there are certainly other reasons for promoting UA. Although it was not the focus of this project, social dimensions are unavoidable in any research or discussion on UA. Given their importance, and the relatively small potential life-cycle environmental benefits, I view most important contributions of UA in many cases as food security/access (not in large amounts of food, but in targeted provisioning of healthy and fresh food), community building, civic engagement, education, recreation, joy, shifting consumption and behaviors, and well-being, among many others. These provide enough reasons to justify UA, and the pretext of reducing environmental impacts of food systems is unnecessary.

Although this perspective appears dismissive of environmental aspects of UA, I argue that environmental assessment is still appropriate. Here, the goals that we proposed for doing UA LCAs attest to their continued relevance. Such assessments can help design and manage UA for low environmental impacts, inform decision/design support for UA planners and practitioners, and help city or project leaders decide which type of UA to promote or develop considering the environmental dimension (among other dimensions hopefully). And of course, since this still an early research topic, continued evaluations of the environmental impacts of UA, testing whether it is impactful at the city level, or under what conditions it could have large benefits compared to the conventional food system, are still necessary.

Appendix

6 Appendix

6.1 Chapter 1: Large-scale characterization of UA

Table 6.1 Frequency of crops grown at different farms

Crop	Frequency	Crop	Frequency	Crop	Frequency
Apples	22	Fava Beans	21	Pears	8
Apricots	1	Fennel	6	Peas	34
Artichoke	7	Figs	7	Peppers- Bell	25
Arugula	17	Garlic	25	Peppers- hot	30
Asparagus	5	Garlic Mustard	2	Physalis	5
Basil	24	Gooseberry	10	Plums	14
Bay	1	Grapes	10	Potato	25
Beet	60	Green Beans	63	Pumpkin	16
Beet- leaves	2	Hazelnuts	2	Purslane	3
Berry (other)	11	Honeydew	3	Quince	4
Black Salsify	1	Hops	1	Radicchio	1
Blackberries	9	Hyssop	3	Radish	43
Blueberries	6	Jerusalem Artichoke	2	Raspberries	26
Broccoli	10	Kale	21	Rhubarb	20
Broccoli Rabe	2	Kiwi	2	Rosemary	12
Butternut Squash	5	Kohlrabi	10	Runner Bean	8
Cabbage	29	Lamb's Lettuce	5	Sage	13
Cantaloupe	1	Lavender	5	Savory	3
Carrot	54	Leek	38	Scallion/green onion	9
Cauliflower	9	Lemon Balm	7	Shadbush	1
Celeriac	25	Lettuce	82	Shallot	4
Celery	11	Loganberries	1	Shepherd's Purse	2
Chamomile	2	Lovage	4	Shiso	1
Chard	30	Marigold	6	Sour Cherry	6
Cherries	12	Marjoram	1	Spinach	22
Chickweed	2	Mint	27	St. Johns Wort	1
Chicory	3	Mirabelle	3	Strawberries	30
Chinese Cabbage (napa cabbage, bok choy)	11	Mizuna Greens (Japanese Greens)	1	Summer Squash	8
Chives	15	Mushroom	2	Sunflower	1

Cilantro	3	Mustard Greens	11	Sweet Potato	3
Collard Greens	7	Nasturtium	3	Thyme	9
Coriander	3	Nettle	3	Tomatillo	1
Corn	9	Okra	6	Tomato	79
Cranberries	1	Onion (red and yellow)	48	Turnip	17
Cucumber	64	Orache	2	Verbena	4
Currant	30	Oregano	8	Vine leaf	2
Daisy	1	Other	23	Violet	1
Dandelion Greens	2	Other (beans)	1	Walnuts	1
Dill	8	Other (edible flowers)	13	Watermelon	5
Dock	8	Other (herbs)	8	Winter Squash	10
Eggplant	21	Parsley	34	Yarrow	1
Elder flower	3	Parsnips	4	Zucchini	42
Endive	2	Peaches	5		

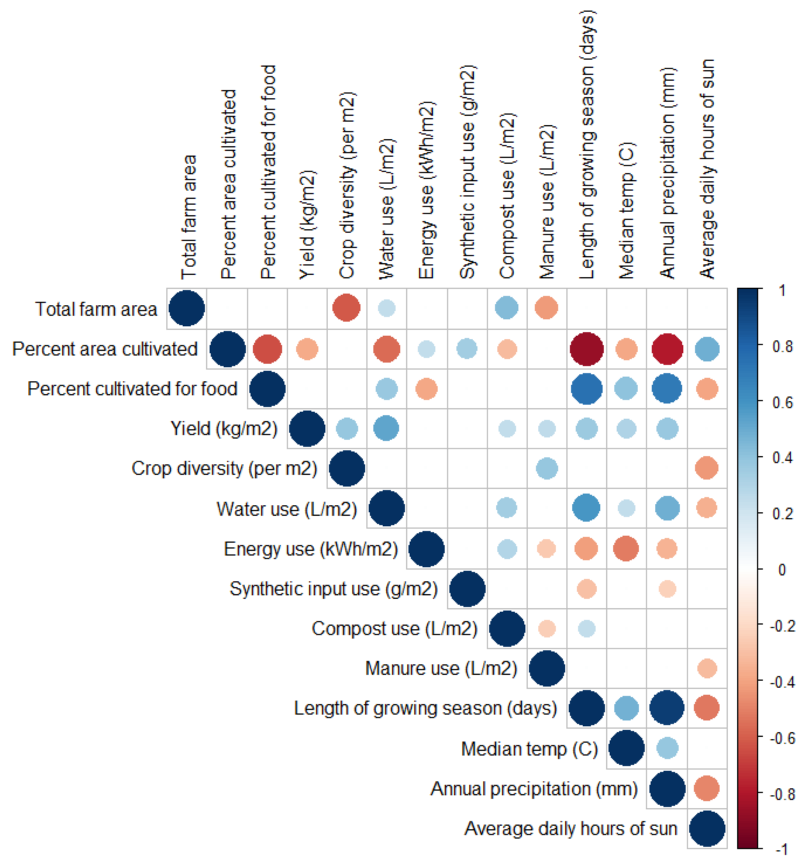

Figure 6.1 A correlation matrix showing the Spearman rank correlations between several variables for the whole dataset. When there is no colored circle at the intersection of two variables, the correlation was not statistically significant (p > 0.05). The color and size of circles show the direction and magnitude of the relationships.

Table 6.2 A correlation table shows the precise values for statistically significant Spearman rank correlations.

row	column	cor	p
Length of growing season (days)	Annual precipitation (mm)	0.946	0
Percent cultivated for food	Length of growing season (days)	0.744	0
Percent cultivated for food	Annual precipitation (mm)	0.705	0
Water use (L/m2)	Length of growing season (days)	0.586	0
Yield (kg/m2)	Water use (L/m2)	0.521	0
Percent area cultivated	Average daily hours of sun	0.482	0
Water use (L/m2)	Annual precipitation (mm)	0.48	0
Length of growing season (days)	Median temp (C)	0.473	0
Total farm area	Compost use (L/m2)	0.431	0
Percent cultivated for food	Median temp (C)	0.403	0
Yield (kg/m2)	Crop diversity (per m2)	0.389	0.001
Median temp (C)	Annual precipitation (mm)	0.384	0.001
Crop diversity (per m2)	Manure use (L/m2)	0.384	0.001
Percent cultivated for food	Water use (L/m2)	0.373	0.001
Yield (kg/m2)	Annual precipitation (mm)	0.372	0.001
Yield (kg/m2)	Length of growing season (days)	0.368	0.001
Water use (L/m2)	Compost use (L/m2)	0.348	0.003
Percent area cultivated	Synthetic input use (g/m2)	0.346	0.003
Yield (kg/m2)	Median temp (C)	0.307	0.009
Energy use (kWh/m2)	Compost use (L/m2)	0.292	0.013
Yield (kg/m2)	Manure use (L/m2)	0.252	0.033
Percent area cultivated	Energy use (kWh/m2)	0.249	0.035
Water use (L/m2)	Median temp (C)	0.249	0.035
Yield (kg/m2)	Compost use (L/m2)	0.248	0.036
Compost use (L/m2)	Length of growing season (days)	0.245	0.038
Total farm area	Water use (L/m2)	0.241	0.041
Synthetic input use (g/m2)	Annual precipitation (mm)	-0.238	0.044
Compost use (L/m2)	Manure use (L/m2)	-0.241	0.042
Energy use (kWh/m2)	Manure use (L/m2)	-0.269	0.022
Synthetic input use (g/m2)	Length of growing season (days)	-0.298	0.011
Percent area cultivated	Compost use (L/m2)	-0.315	0.007
Manure use (L/m2)	Average daily hours of sun	-0.317	0.007
Energy use (kWh/m2)	Annual precipitation (mm)	-0.343	0.003
Water use (L/m2)	Average daily hours of sun	-0.354	0.002
Percent area cultivated	Yield (kg/m2)	-0.37	0.001
Percent cultivated for food	Energy use (kWh/m2)	-0.386	0.001
Percent area cultivated	Median temp (C)	-0.388	0.001
Percent cultivated for food	Average daily hours of sun	-0.394	0.001
Energy use (kWh/m2)	Length of growing season (days)	-0.413	0
Total farm area	Manure use (L/m2)	-0.427	0
Crop diversity (per m2)	Average daily hours of sun	-0.435	0
Annual precipitation (mm)	Average daily hours of sun	-0.489	0
Energy use (kWh/m2)	Median temp (C)	-0.519	0

Length of growing season (days)	Average daily hours of sun	-0.524	0
Percent area cultivated	Water use (L/m2)	-0.569	0
Total farm area	Crop diversity (per m2)	-0.614	0
Percent area cultivated	Percent cultivated for food	-0.64	0
Percent area cultivated	Annual precipitation (mm)	-0.796	0
Percent area cultivated	Length of growing season (days)	-0.868	0

6.2 Chapter 2: How have life cycle assessments of urban agriculture been done, and what have they found?

Table 6.3 Results for keywords search

Keyword search	Number of results
"life cycle assessment" AND "urban agri*" OR "urban farm*" OR "urban garden*"	321
"life cycle" AND "urban agri*" OR "urban farm*" OR "urban garden*"	162
"life cycle" AND "urban agri*" OR "urban farm*"	70
"life cycle assessment" AND "urban agri*" OR "urban farm*"	49
"life cycle analysis" AND "urban agri*" OR "urban farm*" OR "urban garden*"	33
"carbon footprint" AND "urban agri*"	31
"plant factor*" AND "life cycle"	31
"life cycle assessment" AND "urban agri*" OR "urban farm*" OR "urban garden*"	27
"life cycle assessment" AND "building integrated agriculture"	10
"life cycle assessment" AND "urban" AND "hydroponic*"	9
"life cycle assessment" AND "controlled environment agri*"	8
"life cycle assessment" AND "community farm*" OR "community garden*" OR "community agri*"	8
"life cycle assessment" AND "rooftop farm*" OR "rooftop garden*" OR "rooftop agri*"	8
"life cycle assessment" AND "home garden*"	6
"life cycle assessment" AND "urban food system"	5
"life cycle assessment" AND "urban" AND "aquaponic*"	4
"vertical farm*" AND "life cycle"	4
"life cycle assessment" AND "roof-top greenhouse"	2
"carbon account*" AND "urban agri*" OR "urban farm*" OR "urban garden*"	1

6.3 Chapter 4, Part 1: Life cycle assessment of eight diverse urban farms and community gardens

6.3.1 Midpoint characterization impacts from several impact assessment methods, per kilogram of food

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Table 6.4 Environmental Footprint 2.0

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Climate change	kg CO2 eq	0,93717489	1,55456235	0,85313269	1,89211519	1,2035958	1,56876642	1,6147744	3,36611937
Ozone depletion	kg CFC11 eq	6,987E-08	9,4545E-08	4,6738E-08	2,0239E-07	5,8431E-08	2,299E-07	1,0378E-07	2,3049E-07
Ionising radiation, HH	kBq U-235 eq	0,08515186	0,12021474	0,09962591	0,74579407	0,20607281	0,16240343	0,19199909	0,48419525
Photochemical ozone formation, HH	kg NMVOC eq	0,00274958	0,00540524	0,00196315	0,00764884	0,00386046	0,00422113	0,00590374	0,01209727
Respiratory inorganics	disease inc.	4,9058E-08	6,0718E-08	2,3969E-08	7,8733E-08	3,6666E-08	5,4597E-08	5,1371E-08	1,5345E-07
Non-cancer human health effects	CTUh	3,0376E-07	1,607E-07	1,2339E-07	3,8192E-07	1,8382E-07	2,7051E-07	2,4156E-07	1,1595E-06
Cancer human health effects	CTUh	6,4982E-08	2,8878E-08	2,6435E-08	3,0741E-08	4,4248E-08	6,0862E-08	5,426E-08	2,5213E-07
Acidification terrestrial and freshwater	mol H+ eq	0,0052097	0,0071828	0,00300519	0,02306034	0,00650256	0,0067925	0,00829173	0,01826308
Eutrophication freshwater	kg P eq	0,00027629	0,00034166	0,00021788	0,00031931	0,00033414	0,00036375	0,00042557	0,00144905
Eutrophication marine	kg N eq	0,00348793	0,00139248	0,00101643	0,01273806	0,00435769	0,00833974	0,0175563	0,02132289
Eutrophication terrestrial	mol N eq	0,02158552	0,01782086	0,01240051	0,03713022	0,02781928	0,02259581	0,05274856	0,07564017
Ecotoxicity freshwater	CTUe	1,30813131	1,07000146	0,55736422	1,95219724	0,71526216	1,38226589	1,1787977	4,59261896
Land use	Pt	25,9764472	37,5987022	37,9473431	195,553292	56,2576888	34,1080131	73,3071776	110,872684
Water scarcity	m3 depriv.	10,4232148	11,5041411	27,3496931	20,4762269	41,6310776	22,3435863	46,9351993	113,499751
Resource use, energy carriers	MJ	10,6906027	28,188805	11,5822431	34,0640264	14,7669063	28,7008982	14,0144062	41,1763703
Resource use, mineral and metals	kg Sb eq	1,927E-05	5,007E-06	1,7122E-06	5,8884E-06	2,1253E-06	1,6385E-05	5,3012E-06	6,5842E-05
Climate change - fossil	kg CO2 eq	0,89198293	1,54648167	0,84955768	1,91507451	1,2080885	1,56603387	1,60999213	3,34428522
Climate change - biogenic	kg CO2 eq	0,04432023	0,00718475	0,00309026	-0,02415186	-0,00524779	0,00210554	0,0028381	0,01916461
Climate change - land use and transform.	kg CO2 eq	0,00087174	0,00089593	0,00048474	0,00119254	0,00075508	0,00062701	0,00194417	0,00266954

Table 6.5 ReCiPe 2016 Midpoint (H)

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Global warming	kg CO2 eq	0,92552947	1,54263871	0,83803295	1,83111692	1,15850316	1,54287252	1,53090042	3,25852179
Stratospheric ozone depletion	kg CFC11 eq	4,6526E-06	2,4522E-06	1,0153E-06	9,1916E-06	4,2811E-06	5,4945E-06	1,3276E-05	1,5218E-05
Ionizing radiation	kBq Co-60 eq	0,06936321	0,09458451	0,08571622	0,73435121	0,19163502	0,10470836	0,16213818	0,41671146
Ozone formation, Human health	kg NOx eq	0,00177773	0,00355968	0,00108038	0,00352276	0,00162144	0,00225218	0,00224492	0,00602992
Fine particulate matter formation	kg PM2.5 eq	0,00110873	0,00191623	0,00068818	0,00519572	0,00133646	0,00155216	0,00138405	0,00445129
Ozone formation, Terrestrial ecosystems	kg NOx eq	0,001845	0,00373688	0,00111654	0,00363285	0,001681	0,00237432	0,0023171	0,00628442
Terrestrial acidification	kg SO2 eq	0,00303924	0,00468628	0,00164797	0,01535834	0,00337068	0,00413237	0,00336674	0,00994218
Freshwater eutrophication	kg P eq	0,00027631	0,00034172	0,0002179	0,00031941	0,00033418	0,00036379	0,00042562	0,00144925
Marine eutrophication	kg N eq	0,00082674	2,9591E-05	0,00018088	0,00340598	0,00111695	0,00222752	0,00495628	0,00566572
Terrestrial ecotoxicity	kg 1,4-DCB	3,84421908	2,9317817	1,54506321	5,96689191	1,69555588	3,28403468	3,63903236	12,1198096
Freshwater ecotoxicity	kg 1,4-DCB	0,03440189	0,03014339	0,01627716	0,048401	0,02308169	0,04575779	0,0298317	0,14667443
Marine ecotoxicity	kg 1,4-DCB	0,04897719	0,04212028	0,0227604	0,06823836	0,0321507	0,06439748	0,04239791	0,20616646
Human carcinogenic toxicity	kg 1,4-DCB	0,16060744	0,08184707	0,0805807	0,08347119	0,13180411	0,1624001	0,16109068	0,67691873
Human non-carcinogenic toxicity	kg 1,4-DCB	1,0238904	0,75437903	0,36680247	1,16803925	0,55223049	0,99865213	0,73519152	3,69226084
Land use	m2a crop eq	0,11700783	0,13155224	0,1308385	0,88647085	0,20511687	0,11321361	0,29235294	0,41464908
Mineral resource scarcity	kg Cu eq	0,01106232	0,00458353	0,00295423	0,00525617	0,0051629	0,00940494	0,00674444	0,03867101
Fossil resource scarcity	kg oil eq	0,21579428	0,57970237	0,22723064	0,46389071	0,25566604	0,60680628	0,2556516	0,76783992
Water consumption	m3	0,24552942	0,27337395	0,64200477	0,47508686	0,97702471	0,52774589	1,09811575	2,65603387

Table 6.6 TRACI 2.1

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Ozone depletion	kg CFC-11 eq	7,96E-08	1,0805E-07	5,5862E-08	2,6715E-07	7,6092E-08	2,5033E-07	1,2186E-07	2,7706E-07
Global warming	kg CO2 eq	0,89467156	1,50379242	0,82234766	1,7471679	1,10460993	1,50297719	1,41832909	3,08313441
Smog	kg O3 eq	0,04448115	0,08456229	0,03221351	0,11193359	0,05979442	0,06156731	0,09447686	0,18485161
Acidification	kg SO2 eq	0,00375258	0,00589151	0,00202125	0,0156139	0,00372562	0,00497994	0,00414771	0,01204572
Eutrophication	kg N eq	0,00525389	0,00311745	0,0024022	0,01538226	0,00654469	0,01128921	0,02088605	0,03177611
Carcinogenics	CTUh	2,3929E-07	1,2032E-07	1,1626E-07	1,2377E-07	1,902E-07	2,389E-07	2,3338E-07	9,9461E-07
Non carcinogenics	CTUh	5,1486E-07	3,2163E-07	2,034E-07	5,9218E-07	3,0948E-07	4,9018E-07	4,0376E-07	1,9651E-06
Respiratory effects	kg PM2.5 eq	0,00064735	0,0009211	0,00036197	0,00160076	0,00060716	0,00078596	0,00073165	0,00258248
Ecotoxicity	CTUe	11,4111347	9,87867276	5,83246248	18,7923116	8,10170387	17,8120018	10,5852251	51,6494339
Fossil fuel depletion	MJ surplus	0,82780073	2,85496561	0,62226999	2,62257396	0,78043045	2,87815979	1,15976507	3,38435615

Table 6.7 CML-IA baseline V3.05

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Abiotic depletion	kg Sb eq	1,9279E-05	5,0153E-06	1,7164E-06	6,0526E-06	2,135E-06	1,6391E-05	5,3111E-06	6,5863E-05
Abiotic depletion (fossil fuels)	MJ	7,29776568	21,6035929	5,79524968	19,7168353	7,69598924	21,5134327	10,8932648	32,6717115
Global warming (GWP100a)	kg CO2 eq	0,89439443	1,513332	0,82466306	1,74979467	1,11029274	1,50214803	1,41845673	3,10534054
Ozone layer depletion (ODP)	kg CFC-11 eq	6,3182E-08	8,5391E-08	4,5527E-08	2,2332E-07	6,442E-08	1,9216E-07	9,8197E-08	2,268E-07
Human toxicity	kg 1,4-DB eq	0,81760217	0,59204871	0,23033288	0,68879531	0,34993813	0,87056296	0,45925361	2,7859405
Fresh water aquatic ecotox.	kg 1,4-DB eq	0,61626017	0,94928749	0,38310372	1,06623785	0,32099289	0,81752632	0,42352798	2,10950528
Marine aquatic ecotoxicity	kg 1,4-DB eq	1874,17895	4535,05552	1417,89034	1541,90285	781,028168	1334,55445	1022,07729	4465,23758
Terrestrial ecotoxicity	kg 1,4-DB eq	0,00818321	0,00348098	0,00294434	0,09417866	0,00474662	0,00536192	0,00624738	0,02182462
Photochemical oxidation	kg C2H4 eq	0,00018513	0,00030375	0,00011639	0,00090488	0,00029369	0,00028332	0,00029033	0,00085001
Acidification	kg SO2 eq	0,00330894	0,00567174	0,00196332	0,0189237	0,00412259	0,00476506	0,00397813	0,01184245
Eutrophication	kg PO4--- eq	0,00266974	0,00176604	0,0011684	0,00689957	0,00303083	0,00517299	0,00943396	0,01449281

Table 6.8 ILCD 2011 Midpoint+ V1.10

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Climate change	kg CO2 eq	0,85974545	1,45335869	0,77764348	0,13570568	0,8046872	1,50207989	1,05867762	2,63258971
Ozone depletion	kg CFC-11 eq	6,317E-08	8,5357E-08	4,5518E-08	2,2329E-07	6,4408E-08	1,9214E-07	9,818E-08	2,2674E-07
Human toxicity, non-cancer effects	CTUh	5,1486E-07	3,2163E-07	2,034E-07	5,9219E-07	3,0948E-07	4,9018E-07	4,0376E-07	1,9651E-06
Human toxicity, cancer effects	CTUh	2,3928E-07	1,2032E-07	1,1626E-07	1,2377E-07	1,902E-07	2,389E-07	2,3338E-07	9,946E-07
Particulate matter	kg PM2.5 eq	0,00047347	0,00082299	0,00027797	0,0015703	0,00048562	0,00064675	0,00058041	0,00192173
Ionizing radiation HH	kBq U235 eq	0,08515186	0,12021474	0,09962591	0,74579407	0,20607281	0,16240343	0,19199909	0,48419525
Ionizing radiation E (interim)	CTUe	2,7091E-07	4,0848E-07	2,9265E-07	1,5852E-06	5,1578E-07	6,735E-07	5,8094E-07	1,4268E-06
Photochemical ozone formation	kg NMVOC eq	0,00260343	0,00529454	0,00189354	0,00750025	0,00374538	0,00407802	0,00576816	0,01156452
Acidification	molc H+ eq	0,0052097	0,00718279	0,00300519	0,02306033	0,00650256	0,00679249	0,00829172	0,01826308
Terrestrial eutrophication	molc N eq	0,02160353	0,0178278	0,01240384	0,03711958	0,0278203	0,0226091	0,05275679	0,07565888
Freshwater eutrophication	kg P eq	0,00027577	0,00035057	0,00021846	0,00032226	0,00033517	0,00036469	0,00042692	0,00145359
Marine eutrophication	kg N eq	0,00348798	0,00139248	0,00101643	0,01273806	0,00435769	0,00833974	0,0175451	0,02132289
Freshwater ecotoxicity	CTUe	11,4615842	9,91312552	5,85123472	18,8155618	8,13139272	17,8624095	10,6251775	51,845766
Land use	kg C deficit	2,49449316	3,16989583	2,82323853	48,642704	3,77150187	4,83256797	4,07005196	7,12582704
Water resource depletion	m3 water eq	0,04411465	0,05476029	0,11547276	0,09908826	0,17120062	0,10254613	0,1766409	0,42412014
Mineral, fossil & ren resource depletion	kg Sb eq	0,00061254	0,00013784	4,7623E-05	0,00015603	4,1097E-05	0,00051224	0,00014983	0,00207992

6.3.2 Midpoint characterization impacts, PEF, with different functional units

Table 6.9 Impacts per farm per year

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Climate change	kg CO2 eq	6488,99893	12434,8665	1510,54821	898,754714	2548,012307	1162,45592	1488,822	1049,67967
Ozone depletion	kg CFC11 eq	0,00048378	0,00075626	8,2754E-05	9,6134E-05	0,000123699	0,00017035	9,5689E-05	7,1875E-05
Ionising radiation, HH	kBq U-235 eq	589,59147	961,5917	176,396634	354,252182	436,256139	120,34094	177,023157	150,989866
Photochemical ozone formation, HH	kg NMVOC eq	19,0381196	43,2362157	3,47593735	3,63319897	8,172589643	3,12786064	5,44324392	3,77237167
Respiratory inorganics	disease inc.	0,00033967	0,00048568	4,2439E-05	3,7398E-05	7,76213E-05	4,0456E-05	4,7364E-05	4,7853E-05
Non-cancer human health effects	CTUh	0,00210326	0,00128542	0,00021848	0,00018141	0,000389147	0,00020045	0,00022272	0,00036157
Cancer human health effects	CTUh	0,00044993	0,00023099	4,6806E-05	1,4602E-05	9,36734E-05	4,5099E-05	5,0028E-05	7,8623E-05
Acidification terrestrial and freshwater	mol H+ eq	36,0719877	57,4548323	5,32095824	10,9536611	13,76592118	5,03323886	7,64497152	5,69509819
Eutrophication freshwater	kg P eq	1,9130221	2,73289251	0,38578158	0,15167348	0,707373173	0,26953827	0,39237731	0,45186843
Eutrophication marine	kg N eq	24,150393	11,1383513	1,79967368	6,05057629	9,225220559	6,17974712	16,186911	6,64925963
Eutrophication terrestrial	mol N eq	149,458152	142,54813	21,9562117	17,6368559	58,89340759	16,7434966	48,6341713	23,5873834
Ecotoxicity freshwater	CTUe	9057,50121	8558,88821	986,863506	927,293687	1514,210003	1024,25902	1086,85148	1432,1473
Land use	Pt	179860,92	300750,139	67189,1862	92887,8136	119097,5272	25274,0377	67589,2177	34574,1756
Water scarcity	m3 depriv.	72170,3392	92021,0495	48425,0931	9726,2078	88132,99138	16556,5975	43274,2538	35393,3919
Resource use, energy carriers	MJ	74021,7327	225480,842	20507,4038	16180,4125	31261,5407	21267,3655	12921,2825	12840,3049
Resource use, mineral and metals	kg Sb eq	0,13342807	0,04005046	0,00303158	0,002797	0,004499238	0,01214107	0,00488769	0,02053203

Table 6.10 Impacts per total area

Impact category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Climate change	kg CO2 eq	2,49576882	8,34554799	2,15792601	0,50181726	0,402148407	1,3611896	0,42045241	0,43919652
Ozone depletion	kg CFC11 eq	1,8607E-07	5,0756E-07	1,1822E-07	5,3676E-08	1,95231E-08	1,9948E-07	2,7023E-08	3,0073E-08
Ionising radiation, HH	kBq U-235 eq	0,22676595	0,64536356	0,25199519	0,19779575	0,068853557	0,14091445	0,04999242	0,06317568
Photochemical ozone formation, HH	kg NMVOC eq	0,00732235	0,02901759	0,00496562	0,00202859	0,001289866	0,0036626	0,00153721	0,0015784
Respiratory inorganics	disease inc.	1,3064E-07	3,2596E-07	6,0627E-08	2,0881E-08	1,22508E-08	4,7373E-08	1,3376E-08	2,0022E-08
Non-cancer human health effects	CTUh	8,0894E-07	8,627E-07	3,1211E-07	1,0129E-07	6,14184E-08	2,3472E-07	6,2896E-08	1,5129E-07
Cancer human health effects	CTUh	1,7305E-07	1,5503E-07	6,6865E-08	8,1529E-09	1,47843E-08	5,2809E-08	1,4128E-08	3,2896E-08
Acidification terrestrial and freshwater	mol H+ eq	0,01387384	0,03856029	0,00760137	0,00611595	0,002172652	0,00589372	0,00215899	0,00238289
Eutrophication freshwater	kg P eq	0,00073578	0,00183416	0,00055112	8,4686E-05	0,000111643	0,00031562	0,00011081	0,00018907
Eutrophication marine	kg N eq	0,00928861	0,0074754	0,00257096	0,00337832	0,001456001	0,00723624	0,00457128	0,00278212
Eutrophication terrestrial	mol N eq	0,0574839	0,09566989	0,03136602	0,00984749	0,009295045	0,01960597	0,01373459	0,0098692
Ecotoxicity freshwater	CTUe	3,48365431	5,74422028	1,40980501	0,51775192	0,238985165	1,19936654	0,30693349	0,59922481
Land use	Pt	69,177277	201,845731	95,9845518	51,8636592	18,79695821	29,5948919	19,0876074	14,4661823
Water scarcity	m3 depriv.	27,7578228	61,7590936	69,1787044	5,43060179	13,90987869	19,3871165	12,2209132	14,8089506
Resource use, energy carriers	MJ	28,4698972	151,329424	29,2962912	9,03428952	4,933955287	24,9032383	3,64904901	5,37251249
Resource use, mineral and metals	kg Sb eq	5,1318E-05	2,688E-05	4,3308E-06	1,5617E-06	7,10107E-07	1,4217E-05	1,3803E-06	8,5908E-06

Table 6.11 Impacts per green area

Impact Category	Unit	FR1	FR2	FR3	FR4	US1	US2	US3	US4
Climate change	kg CO2 eq	25,6482171	41,72774	3,80490732	3,28591302	2,513879226	1,90566544	1,56924695	1,76535431
Ozone depletion	kg CFC11 eq	1,9122E-06	2,5378E-06	2,0845E-07	3,5147E-07	1,22041E-07	2,7927E-07	1,0086E-07	1,2088E-07
Ionising radiation, HH	kBq U-235 eq	2,33040107	3,22681779	0,44432402	1,29517191	0,430412068	0,19728023	0,1865858	0,25393519
Photochemical ozone formation, HH	kg NMVOC eq	0,07524948	0,14508797	0,00875551	0,01328324	0,00806311	0,00512764	0,00573728	0,00634439
Respiratory inorganics	disease inc.	1,3426E-06	1,6298E-06	1,069E-07	1,3673E-07	7,65814E-08	6,6322E-08	4,9922E-08	8,0479E-08
Non-cancer human health effects	CTUh	8,3133E-06	4,3135E-06	5,5032E-07	6,6325E-07	3,83934E-07	3,2861E-07	2,3475E-07	6,081E-07
Cancer human health effects	CTUh	1,7784E-06	7,7514E-07	1,179E-07	5,3385E-08	9,24185E-08	7,3933E-08	5,273E-08	1,3223E-07
Acidification terrestrial and freshwater	mol H+ eq	0,14257703	0,19280145	0,01340292	0,04004739	0,013581513	0,00825121	0,00805795	0,00957803
Eutrophication freshwater	kg P eq	0,00756135	0,00917078	0,00097174	0,00055453	0,000697897	0,00044187	0,00041357	0,00075995
Eutrophication marine	kg N eq	0,0954561	0,03737702	0,00453318	0,02212135	0,00910164	0,01013073	0,01706131	0,01118274
Eutrophication terrestrial	mol N eq	0,59074368	0,47834943	0,05530532	0,06448164	0,058104474	0,02744836	0,05126135	0,03966933
Ecotoxicity freshwater	CTUe	35,8004001	28,7211014	2,48580228	3,39025359	1,493925701	1,67911316	1,14556232	2,40858947
Land use	Pt	710,912728	1009,22865	169,242283	339,604646	117,502101	41,4328487	71,2403322	58,1469486
Water scarcity	m3 depriv.	285,258258	308,795468	121,977564	35,5597276	86,95236503	27,141963	45,6118937	59,5247089
Resource use, energy carriers	MJ	292,576019	756,64712	51,6559291	59,1567725	30,84276223	34,8645337	13,6192797	21,5948618
Resource use, mineral and metals	kg Sb eq	0,00052738	0,0001344	7,6362E-06	1,0226E-05	4,43897E-06	1,9903E-05	5,1517E-06	3,4531E-05

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6.3.3 Data collection by farm

6.3.3.1 *FR1*

Dates: September 2019-August 2020

Harvest: There was a unique method for calculating harvest data at FR1 because it is a commercial farm where customers pay to rent a plot, which farmers take care of, and customers come about once per week to harvest. Since it was unreasonable to ask all customers to weigh their produce, we took data from the three control plots and extrapolated it to the 300 customer plots. The control plots are managed by the farmers, and harvest is done by the farmers, so weighing and recording is more feasible. All plots have exactly the same layout, same crops planted, and receive the same amount of water and fertilizer through a farm-wide fertigation system. Produce was harvested from April through December.

Harvest data were available for 17 months, and total harvest ranged from 6924-9718 kg per year, with an average of 8744 kg. We used the period with a harvest of 6924 kg because that period had the most reliable irrigation data.

Water: We used the water bill for the entire rooftop area to estimate the water used. Since the farm occupies 87% of the cultivated roof area, we allocated 87% of the amount in the water bill to the farm.

Compost and other supplies: The main inputs here were liquid organic fertilizers. The farmers provided technical specifications sheets for each fertilizer, which was provided to them by the fertilizer supplier with specialized instructions on how much to use based on their substrate composition and crop choices. Additionally, the farmers estimated how much of each fertilizer they used based on the frequency with which they use up a container. We calculated the annual fertilizer use based on both of these estimates, and used the average of these calculations for the LCA. For all fertilizers, the estimates varied by an average of 18%.

Delivery and packaging: There were no delivery or packaging because customers come to the farm. In a sensitivity analysis we aimed to estimate the customer travel to the farm, but very little data were available and no customers responded to our survey on the matter. Our only data point was that 2/3 of customers live within 5 km of the farm, and 1/3 live between 6-19 km from the farm. We proceeded with an estimation that for each of the 300 plots, someone makes a trip to the farm one time per week, and half of those trips are on the way for another trip and do not come with impacts. The weighted average distance to the farm was 5.79 km. Based on the farm's suburban position and rather inconvenient access to public transport stops, we estimated that 50% of trips would be done by car, 30% by metro, and 20% by bike.

6.3.3.2 *FR2*

Dates: January 2020-December 2020

Harvest: Harvest was recorded by farmers as produce was picked. Because this is a commercial farm, with only a handful of employees working, there was a very low likelihood of harvest going unrecorded. Produce was harvested from March through November.

Water: Because this was a new farm, and we used data that the farm had already collected in their first year of operation, no water data were available. We assigned the same water use per kilogram (m^3/kg) as was measured for FR1, since they were assumed to have similarly efficient water use. FR1 used precise drip fertigation, had very little issues with leaks, and ultimately had much lower water use per kilogram than other farms. FR2 was assumed to also be water efficient because of their hydroponics and aeroponics technologies. It is likely that they even had lower water use than FR1, because water was recycled in the hydroponics system, but with a lack of data we made conservative estimates.

Compost and supplies: The farmers provided information about what supplies were used and how much. This included fertilizers and substrate pods. As a commercial farm, they had purchase records and the information was readily available.

Delivery and packaging: As a commercial organization, FR2 had information about how much produce went to each client, and general knowledge of the frequency of delivery. They reported the mode of delivery for each sale: mostly walking, but some delivery was done by car. They also provided the materials and amount of packaging because they regularly ordered the necessary crates and boxes.

6.3.3.3 *FR3*

Dates: January 2019-December 2019

Harvest: Harvest was recorded in a notebook by farmers. They weighed the produce and recorded the crop and weight. Very little to no produce was unrecorded.

Here we used data from 2019 when 1771 kg were harvested. Data were also available from 2018 and 2020, although the farm changed in size as new parcels were developed. In 2018 harvest was 957 kg, and in 2020 was 2847 kg. In 2019 produce was harvested from May through November, although in other years there were also small harvests from January through April.

Water: Water use was counted using water meters installed throughout the farm for this research project. Four water meters were installed, covering the drip irrigation (the main water use on the farm), hose use, and sink use for washing produce. Farmers recorded the water meter readings approximately monthly: the average duration between readings was 28 days. Leaks were a large problem at this site.

Compost and other supplies: Compost was the only input used here and was delivered and applied once per year. The farm consistently used and made the same amount of compost for several years, so we used that amount for the LCA.

Delivery and packaging: All delivery was done on foot or by bike to the same location less than 1 km away. This was an office building, where produce boxes were dropped off to employees working in the building. We assumed that plastic crates were used for delivery and reused 50 times, as described in Sanyé-Mengual et al. (2013).

6.3.3.4 *FR4*

Dates: May 2019-April 2020

Harvest: Harvest data were collected in a notebook by farmers and students. When something was picked, farmers or students weighed it and recorded the crop type, weight, parcel, and destination. Farmers reported some products going unrecorded, such as strawberries that students mostly ate directly from the plant without recording, although this was estimated to be relatively unimportant.

Harvest data were available from fifteen months, therefore four possible 12-month combinations. The total harvest ranged from 459-506 kg harvested in a 12-month period, and we chose an intermediate period with 475 kg harvested. Produce was harvested year-round, although was less important from January through April.

Water: Water use was counted using water meters installed throughout the farm for this research project. Four water meters were installed, covering the drip irrigation (the main water use on the farm), hose use, and sink use for washing produce. Farmers recorded the water meter readings approximately month: the average duration between readings was 26 days. Researchers collected the readings during monthly farm visits.

Compost and other supplies: Information about use of compost and other supplies was collected during monthly farm visits by researchers and casual interviews with the farmers. They estimated the amount of supplies delivered to the farm since the last visit, and the amount of supplies used. Estimated of supply use were less reliable than supply delivery, so we used supply delivery estimates as the main data source. The source of supplies, distance from the farm, and frequency of delivery were also accounted for.

Delivery and packaging: Destination of produce was recorded in the harvest notebook. All delivery was done on foot or by bike. No packaging was necessary.

6.3.3.5 *US1*

Dates: July 2020-June 2021

Harvest: The community farm is open to the public and harvesting is done by a mix of the public and volunteers, with more or less experience with the farm. People are instructed to weigh and record any produce that they harvest, but this is not always done. Various farmers who spend a lot of time at the farm estimated that 30-50% of harvest is unrecorded. Crops were harvested year-round.

There was large variability in the harvest data across months and years. We used data from the only 12-month period where we also had compost and water data (described below). From January 2019 to May 2021, the 12-month total harvest ranged from 2053 to 3456 kg, with an average of 2673 kg. During the 12 months of our study, the harvest was 2117 kg. Past data from farm records show that in 2015 and 2016 the harvests were 5231 and 5825 kg. Farmers suggested that reasons for the large difference in yield could be that in the past there was a dedicated farm manager who made the growing more efficient, and who reminded people more to record the harvest. They also estimated that they used much more compost in the past.

Water: To collect water use data we worked with the one volunteer who was charge of the water at this community farm. We installed eight water meters across the farm, and the farmer recorded water use approximately every two weeks (average duration of 15 days). The farmer filled out a Google Form with the water meter readings to communicate the values to researchers.

Compost and other supplies: The supplies used here were mushroom compost, purchased compost, and farm compost. We collected this data with the volunteer in charge of the composting operations at the farm. Mushroom compost was delivered weekly for several months, so the volunteer told us how many times it was delivered and the typical quantity.

Purchased compost amounts were available from purchase records. Farm compost was especially important at this farm because they make a lot of compost, and they collect organic waste from elsewhere to compost at the farm. The volunteer regularly emailed researchers, about every two weeks, with information on what waste was brought to the farm, and how many piles of compost had been applied. We measured the volume of several finished piles of compost, and used the average volume to calculate how much compost had been created and used.

Delivery and packaging: Since community members came to the farm to pick up or harvest produce, no delivery or packaging processes were assigned in the baseline scenario. However, to estimate this for the sensitivity analysis, we used the destination information provided sometimes with harvest records. This information included the zip code of an individual or group who came to the farm, a note stating it was distributed at the weekly farm stand with no information as to who took it, or the name of a community organization that the produce was donated to. We assumed that half of the trips to the farm were done during another trip and were not assigned impacts, and 80% of trips were done by car, 10% by bike, and 10% by bus.

6.3.3.6 *US2*

Dates: January 2020-December 2020

Harvest: Weight and type of crop was recorded by researchers and student volunteers. The records were estimated to be of moderate quality, with some harvests likely unrecorded, although not a substantial amount. In 2020 crops were harvested only from June through December, and in 2019 from May through October. Although growing could occur year-round in this climate, limits on time and effort from the main researcher and farmer at the site meant that they could not manage year-round cultivation.

Data were available from 2019 and 2020. In 2020, total harvest was 657 kg, and in 2019 it was 825 kg. This was the only case where we used the average harvest from multiple years, because we were guaranteed by the researchers at the site that all practices were the same over the two years. The final harvest amount used in the LCA was 741 kg.

Water: Water data were available from previously installed water meters for the research project.

Compost and other supplies: The amount of compost used per year was already known because it had been tracked for another research project. Feather meal and oyster shell use was already known, although this was very small (about 1.5 and 0.5 liters for each input).

Delivery and packaging: Destination of the produce was recorded in the harvest file. Most produce was delivered to local community groups 1-15 km away from the farm. Thin produce bags were used as packaging.

6.3.3.7 *US3*

Dates: July 2020-June 2021

Harvest: This is a U-pick farm where customers come and harvest their own produce. They grow many uncommon herbs and vegetables, and the farmers usually spend around 10 minutes with each new customer showing them around the farm, telling them what they're growing, and letting them taste many herbs and edible flowers. Produce is priced by the bag (half bag or full bag), so normally there is no weighing of produce and farmers do not know how much they grow. They record the revenue, number of customers, and what crops were harvested (but no breakdown of how much of each crop). For this study, farmers agreed to weigh the bags approximately once per week, in order to not disrupt the customer experience and take too much time. Using the matched daily revenue and weight values, we created a regression model that predicted weight harvested on other days using only the revenue.

We split up the data to make multiple models based on the different seasons because the type of crops harvested varied by the season: in winter lighter products were harvested like herbs and kale, and in the late summer the heaviest crops were harvested like squash and tomato. A non-linear regression model was used with a power equation for four seasons. The seasons were determined with input from the farmer based on when their crop choices changed, and based on the number of data points we had for each season. The seasons were early summer from June 5th-July 10th (we had weight measures for almost every day here), late summer from July 11th-October 31st, winter from November 1st to March 6th, and spring from March 7th-June 4th. The regression models were done in R on the log of the revenue and harvest data. The resulting p values, adjusted R^2 values, and equations are shown in table X. The values used (not log transformed) are shown in Figure 6.1. Revenue data was adjusted to include only sales of crops harvested, and not compost and potted plants. The estimated total harvest was 923 kgs.

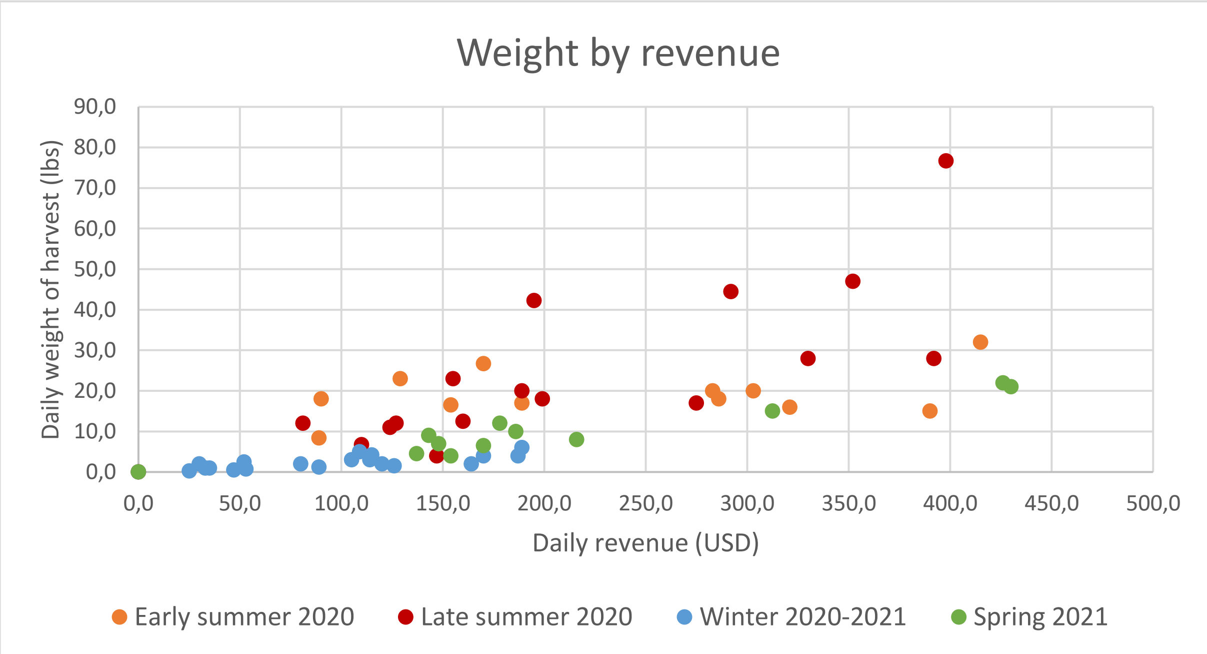

Figure 6.2 Paired measurements for daily weight of harvest and revenue measured for US3, across different seasons. These pairs were used in a regression analysis to predict harvest for other days when it wasn't measured based on the revenue.

Water: We used the water meter bills for the farm to obtain the amount of water used. This measured water used at the entire site, which was appropriate because there were no other activities at the site.

Compost and other supplies: There was a very consistent crop rotation, bed preparation schedule and bed preparation practice at the farm. We extrapolated the amount of compost applied in one bed preparation to all crop beds for one year to find the amount of compost used. The amount of purchased compost was available through purchase records, and we subtracted this amount from the total compost applied to find the on-farm compost production. The farm also used a substantial amount of feather meal, and smaller amounts of oyster shells, alfalfa meal, and kelp meal. The quantities of these supplies used were available in purchase records.

Delivery and packaging: As a U-pick farm, US3 did not deliver products, and instead customers came to the farm. In the baseline scenario, no delivery impacts were given to US3. In the sensitivity analysis including customer travel to the farm, we estimated transport distances based on some days where the zip code or city of customers was tracked. This data was available for two months, and we extrapolated the distance of customers over the entire year, using the daily recorded number of customers. We assumed that all travel was done by car because the farmer told US4 that nearly all customers came by car, and the farm is not accessible by public transport, and is on a very steep hill so is rather inaccessible by bike.

6.3.3.8 *US4*

Dates: July 2020-June 2021

Harvest: Similar to FR4, harvest data were collected in a notebook by farmers and students. The type of crop and its weight were recorded. The main farmer estimated that some harvest was unrecorded, but was unsure of how much. Produce was harvested year-round.

Harvest data were available for 21 months of operation, and the 12-month harvest ranged from 225-319 kg with an average of 248 kg. We used the 12-month period with 312 kg because that's when water data were available.

Water: Water use was counted using water meters installed throughout the farm for this research project. Four water meters were installed, covering the drip irrigation (the main water use on the farm), hose use, and sink use for washing produce. Farmers sent photos of the water meter readings to the researchers approximately once per month: the average duration between readings was 27 days. Regular leaks were a common issue at this farm.

Compost and other supplies: Purchased compost amounts were available from purchase records, and on-farm compost amounts were estimated using the number of wheelbarrows of compost applied every few months. Other inputs, including kelp and fish emulsion fertilizers and feather meal, were estimated in annual amounts because small quantities were used.

Delivery and packaging: Most of the produce was distributed in produce boxes at a weekly drop off point 5 km away from the farm, for 40 weeks of the year. Thin plastic produce bags were used as packaging, and the farmers provided the amount that they ordered every year.

6.3.4 Nitrogen inputs

Table 6.13 The amount of nitrogen applied per farm, in total amount per year, amount per area, and amount per kilogram of crop grown.

Farm	kg N applied	kg N/ha	kg N/kg crop
FR1	75	2964	0.011

FR2	17	577	0.002
FR4	34	1383	0.072
FR3	5	136	0.003
US1	35	398	0.017
US2	31	500	0.041
US3	131	2059	0.142
US4	36	655	0.116
Average	46	1084	0.05

6.3.5 Nitrogen losses

We accounted for three fates of applied nitrogen: direct emissions as N_2O, runoff at NO_3, and indirect emissions of N_2O from the NO_3.

First, we calculated the amount of nitrogen applied on the farms. The amount of nitrogen was calculated using a nitrogen content 0.9% dry mass for compost (Amlinger et al., 2008; Chia et al., 2020; Khater, 2015; Siedt et al., 2021), and 16% for feather meal (which was a major input for US3) (Hadas and Kautsky, 1994). For mineral and organic liquid fertilizers used at FR1 and FR2, technical sheets were provided with the nitrogen contents of each fertilizer, so the nitrogen applied was directly calculated. We used the IPCC Tier 1 rate of 0.5% of organic N applied in soils is emitted as N in N_2O (IPCC, 2019), and multiplied by 1.57 to obtain the amount of N_2O emitted. We used a rate of 24% of organic N applied in soils leached as N (IPCC, 2019), 99% remains in waterways, and multiplied by 4.43 to obtain the amount of NO_3 leached to waterways. Finally, for indirect N_2O emissions, we assumed that 1% of the leached N is volatilized, and multiplied by 1.57 to obtain the amount of N_2O emitted. For the mineral fertilizers used in hydroponics systems, we used a rate of 0.80% of N applied in hydroponics is emitted as N in N_2O (Llorach-Massana et al., 2017b). The leaching rates were the same for FR1, but for FR2 we assumed that was no leaching due to the closed hydroponics system.

6.3.6 Compost system modeling, emissions and parameters

Compost was a major input at most farms, and the system modeling decisions and emissions of composting are highly variable. Plus, some important parameters were chosen to treat compost in the LCI: the ratio of waste to compost, the bulk density of fresh compost, and the moisture content.

Industrial/municipal composting is an example of open-loop recycling, where the secondary function (making compost) is used in different systems for different purposes than the primary function (organic waste treatment). Since a valuable product is made during composting, the secondary good (compost) can be seen as a co-product of the first system (the one that generated organic waste) (European Commission, 2010b). In this case, economic allocation based on the market value of waste treatment vs compost production is appropriate to distribute impacts. According to Pepin (2022), 92.8% of revenue at composting facilities comes from fees for dumping organic waste, and 7.2% comes from purchases of compost. We allocated compost 7.2% of the impacts from the composting process.

The main emissions from composting are greenhouse gases: methane (CH_4) and dinitrous monoxide (N_2O). These gasses are emitted by bacteria decomposing the organic matter in the composting process, when they are under anaerobic conditions (lacking oxygen). Emission rates are affected by compost moisture content, turning of compost piles, other forced aeration practices, feedstock material, and choice of bulking agents (Pardo et al., 2015). In Chapter 3, we summarized the emissions in some of the major literature sources for other UA LCAs, plus LCA databases. We selected the emissions reported in the review and meta-analysis by Nordahl et al. (2022), assuming a mix of 70% yard waste and 30% organic fraction of municipal solid waste. Values for NH_3 and VOC emissions also came from Nordahl et al. (2022). In a sensitivity analysis, we calculated our results using the greenhouse gas emission values from Ecoinvent, which came from Edelmann and Schleiss (1999).

The ratio of waste to compost was an essential parameter because most composting LCAs were available based on impacts per kilogram of waste treated. Conversion factors from mass of wet waste treated to mass of compost created are variable and depend on several factors, including the type of feedstock (i.e. the waste being treated) and the compost management practices. We used a waste to compost ratio of 2 (i.e. 2 kilograms of waste generate 1 kg of compost) based on values found in the literature (Andersen et al., 2012, 2010; APESA et al., 2015; Breitenbeck and Schellinger, 2004; Martínez-Blanco et al., 2010). In a sensitivity analysis, we used a ratio of 1.5.

Compost density was another critical parameter because the amount of compost used at farms was usually given in volume, and LCAs for compost were available in mass. We found many measurements of compost fresh bulk density, ranging from 300 to 900 kg/m^3 (Agnew and Leonard, 2003; Colón et al., 2010; Jain et al., 2019; Khater, 2015; Schaub-Szabo and Leonard, 1999). For the baseline scenarios a value of 450 kg/m^3 was chosen, but we evaluated other densities in sensitivity analyses. In a sensitivity analysis, we used 350 kg/m^3.

Compost moisture content was used to calculated the dry mass of compost, which was necessary for calculating nitrogen and carbon content, as described below. In our literature search, values ranged from 23-75% (Agnew and Leonard, 2003; Andersen et al., 2011; Jain et al., 2019; Khater, 2015; Martínez-Blanco et al., 2010; Schaub-Szabo and Leonard, 1999). We used an average value of 50% moisture content.

6.3.7 Soil carbon sequestration

Estimates of long-term soil carbon sequestration have very high uncertainty, and in some cases are not recommended to include in LCAs ((Christensen et al., 2018; Nordahl et al., 2022; Seufert and Ramankutty, 2017). For this reason, soil carbon sequestration is not included in the main results, and rather was a supplementary result.

We first calculated the dry weight of compost applied, using the moisture content described above. We assumed a common organic carbon content for all composts of 12% of dry mass (Amlinger et al., 2008; Andersen et al., 2011; Chia et al., 2020; Houot et al., 2014; Jain et al., 2019; Khater, 2015; Siedt et al., 2021). The largest uncertainty in long term carbon sequestration seems to lie in the estimate of the fraction of carbon sequestered in the long-term. We found values in the literature ranging from 2-16% of applied carbon sequestered in the long-term, and used an intermediate value of 10% (Boldrin et al., 2009; Favoino and Hogg, 2008; Martínez-Blanco et al., 2013; Tonini et al., 2020; Vaneeckhaute et al., 2018).

6.3.8 Infrastructure

Infrastructure objects were measured by researchers at all farms except for FR2, where measurements were provided by farmers. Lifetimes for infrastructure were determined based on the expected lifetime of a material or of the object (i.e. the lifetime of drip tape is limited by the nature of the object rather than the integrity of the plastic). The lifetimes for the most common materials and objects are in table 3. Delivery of infrastructure to the farm was included, although farmers often didn't know the suppliers, so when this information wasn't available, we assumed it came from the nearest large hardware store. End of life waste treatment was also determined by either the material or the object. For example, irrigation pipes are usually made of recyclable plastic, but since they often end up full of soil, they are not usually recycled in reality. In an alternative scenario, we doubled the lifetime of all infrastructure.

Table 6.14 Lifetime of infrastructure objects or materials, and the assumed waste treatment.

Material/object	Lifetime	Waste treatment
Substrate	15	None
Wood	10	Recycle
Netting	5	Waste mix
Geotextile	10	Waste mix
Irrigation pipes	5	Waste mix
Steel	15	Recycle

6.3.9 Seedlings

We accounted for purchased seedlings for FR1, FR2, FR3, US1 and US2, because all plants at these farms came from seedlings started off of the farm. Other farms started all or nearly all plants from seeds at the farm. Purchased seed packages were not included in the LCA.

FR1 had detailed records of the number of seedlings they ordered: 72,301 for one year. We calculated the seedlings per kilogram of food grown there, and applied the same ratio to calculate the number of seedlings needed at the other farms to reach their total harvest. We adjusted the harvest at the other farms to exclude crops that are usually seeded (such as lettuce, collard greens, chard, radishes, and beets), because it was assumed that these were directly seeded in the plots at the farm. Our calculations for US2 corresponded with the farmer's estimate of using 5,000-10,000 seedlings per year, where we calculated they used 7,839. US2 was the only farm were such an estimate was made. We used the processes in Ecoinvent for tomato seedlings. For FR1, because even their "light" seedling plants came from off-farm (such as lettuce, chard, radishes…) we used a mix of tomato seedlings ("heavy") and onion seedlings ("light").

6.3.10 Irrigation

We used the European tap water mix process in Ecoinvent (Tap water {RER}| market group for | Cut-off, S) despite the locations of the farms. This affects the water scarcity impacts calculated by the AWARE method (Boulay et al., 2018). This impact assessment method accounts for not only the amount of water used, but also the availability of water in a given location. The California farms would be expected to have a much higher water scarcity impact due to the contemporary drought conditions there, especially during 2012-2015 (Griffin and Anchukaitis, 2014; Swain et al., 2018). However, the AWARE method is limited and would not have captured this, because it uses water availability data from 1960-2010. The characterization factors for the coastal Bay Area, California watershed are surprisingly very similar to those of the Ile de France (Paris region) water shed (https://wulca-waterlca.org/aware/download-aware-factors/). Plus, we wanted to evaluate more average, representative scenarios, and account for current drought issues in California would have largely affected the water results.

6.3.11 Irrigation electricity

The only electricity directly used on the farms was for irrigation timers and pumps. We used data based on the rooftop, open-air system in Goldstein et al. (2016b), where 161.5 MJ was used for this purpose for 423 m^2 growing area. We applied the same energy use per m^2 to all farms here. This was a rough approximation, but because it came with very low impacts in Goldstein et al. (2016b) and in our preliminary results, we deemed the level of accuracy satisfactory. This energy use was included in the “Irrigation” process of the LCAs.

6.4 Chapter 4, Part 2: Life cycle assessment of a circular, urban mushroom farm

6.4.1 Life cycle inventory: Ecoinvent codes

Table 6.15 The corresponding Ecoinvent process names are shown below for each general process that was given in the life cycle inventory (Table 4.3) of chapter 4, part 2.

Life cycle stage	Input	Material	Ecoinvent code
Substrate materials	Coffee ground	Transport, 3.7-7.5 ton lorry (EURO 5)	Transport, freight, lorry 3.5-7.5 metric ton, EURO5 {GLO}| market for | Alloc Rec, U
	Wood chips	Wood chips, as a byproduct	Wood chips, wet, measured as dry mass {RER}| market for | Alloc Rec, U
		Transport, 3.7-7.5 ton lorry (EURO 5)	Transport, freight, lorry 3.5-7.5 metric ton, EURO5 {GLO}| market for | Alloc Rec, U
	$CaCO_3$	Lime	Lime {GLO}| market for | Alloc Rec, U

		Transport, 3.7-7.5 ton lorry (EURO 5)	Transport, freight, lorry 3.5-7.5 metric ton, EURO5 {GLO}| market for | Alloc Rec, U
	Mycelium	Mycelium inoculated rye seeds	Inventory taken from (Leiva et al., 2015b)
		Transport, 3.7-7.5 ton lorry (EURO 5)	Transport, freight, lorry 3.5-7.5 metric ton, EURO5 {GLO}| market for | Alloc Rec, U
		Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Water	Tap water	Tap water {Europe without Switzerland}| market for | Alloc Rec, U
Substrate transformation	Air purification	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Conveyor belt	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Substrate mixing	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Substrate cooling	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Sterilization: Gas	Sour gas, global average	Sour gas, burned in gas turbine {GLO}| market for | Alloc Rec, U
	Sterilization: Water	Tap water	Tap water {Europe without Switzerland}| market for | Alloc Rec, U
	Plastic bags	Polyethylene, low density	Polyethylene, linear low density, granulate {GLO}| market for | Alloc Rec, U
	Air purification	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
Cultivation	Air temperature regulation	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Humidifier	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U

	LED lighting	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Ventilation	Electricity, French grid	Electricity, medium voltage {FR}| market for | Alloc Rec, U
	Water	Tap water	Tap water {Europe without Switzerland}| market for | Alloc Rec, U
	Steel racks	Steel, low-alloyed	Steel, low-alloyed {GLO}| market for | Alloc Rec, U
	Sanitary materials	Polypropylene	Polypropylene, granulate {GLO}| market for | Alloc Rec, U
		Polyethylene, low density	Polyethylene, low density, granulate {GLO}| market for | Alloc Rec, U
		Polyethylene, high density	Polyethylene, high density, granulate {GLO}| market for | Alloc Rec, U
		Synthetic rubber	Synthetic rubber {GLO}| market for | Alloc Rec, U
Packaging and delivery	Wood crates	Plywood, for indoor use	Plywood, for indoor use {RER}| market for | Alloc Rec, U
		Transport, 3.7-7.5 ton lorry (EURO 5)	Transport, freight, lorry 3.5-7.5 metric ton, EURO5 {GLO}| market for | Alloc Rec, U
	Delivery	Transport, passenger car, large size, diesel (EURO 5)	Transport, passenger car, large size, diesel, EURO 5 {GLO}| market for | Alloc Rec, U

6.4.2 Carbon Sequestration Calculations

In an experiment described in Medina et al. (2012), SMS was applied to soils and the change in SOC was measured. The SMS was a mix of equal parts *Pleurotus* and *Agaricus bisporus* substrate, and was mixed in to a soil depth of 0.3 m. After 126 days, the soil organic carbon content increased by 1.1 g C_{org}/ kg soil. Assuming a bulk density of 1.2 kg soil/L soil, the C_{org} sequestered per area of soil is:

$$\frac{0.0011\ kg\ C_{org}}{kg\ soil} \times \frac{1.2\ kg\ soil}{L\ soil} \times \frac{1000\ L\ soil}{m^3\ soil} \times 0.3\ m\ soil = \frac{0.396\ kg\ C_{org}\ sequestered}{m^2\ soil}$$

The input rate of SMS was 8.5 kg SMS/m^2 of soil, and the SMS had a C_{org} content of 35.1%. This is used to find the amount of C_{org} applied by SMS to the soil:

$$\frac{8.5\ kg\ SMS}{m^2} \times \frac{0.351\ kg\ C_{org}}{kg\ SMS} = \frac{2.98\ kg\ C_{org}\ applied\ from\ SMS}{m^2\ soil}$$

The percent of C_{org} applied that was sequestered over 126 days was:

$$\frac{0.396\ \frac{kg\ C_{org}\ sequestered}{m^2\ soil}}{2.98\ \frac{kg\ C_{org}\ applied\ from\ SMS}{m^2\ soil}} = 13.27\%\ kg\ C_{org}\ applied\ from\ SMS$$

This sequestration factor was used to estimate the amount of C that would be sequestered from the mass of SMS produced at the farm during the study period. Standard agronomic soil testing showed that the farm's SMS had an organic matter content of 86.0%, which is similar to the organic matter content of *Pleurotus* SMS measured by Paredes et al. (2009) of 86.9%. Therefore, the organic carbon content of SMS was taken from Paredes et al., and was 43.2%. During the study period, 7,780 kg of SMS were produced, so the kg of organic carbon in the SMS was:

$$7{,}870\ kg\ SMS \times \frac{0.432\ kg\ C_{org}}{kg\ SMS} = 3{,}399\ kg\ C_{org}\ in\ SMS$$

And the amount of organic carbon from SMS that was sequestered per kilogram of mushroom produced was:

$$3{,}399\ kg\ C_{org}\ in\ SMS \times 0.1327\ \frac{kg\ C_{org}\ sequestered}{kg\ C_{org}\ in\ SMS} \div 1253\ kg\ mushrooms$$
$$= 0.36\ \frac{kg\ C_{org}\ sequestered}{kg\ mushroom}$$

This is converted to atmospheric CO_2 equivalents using the atomic mass of CO_2:

$$0.36\ \frac{kg\ C_{org}}{kg\ mushroom} \times 3.67\ \frac{kg\ CO_2}{kg\ C} = 1.32\ \frac{kg\ CO_2\ eq.}{kg\ mushroom}$$

6.4.3 Midpoint characterization impacts from several impact assessment methods

Table 6.16 Characterization Results: Environmental Footprint 2.0

Impact category	Unit	Total
Climate change (w/ C seq.)	kg CO2 eq	2.99058759
Climate change (w/out C seq.)	kg CO2 eq	3.18282972
Ozone depletion	kg CFC11 eq	6.4615E-07
Ionising radiation, HH	kBq U-235 eq	4.82050264
Photochemical ozone formation, HH	kg NMVOC eq	0.0189625
Respiratory inorganics	disease inc.	2.4882E-07
Non-cancer human health effects	CTUh	4.2294E-07
Cancer human health effects	CTUh	4.5579E-08
Acidification terrestrial and freshwater	mol H+ eq	0.13990633
Eutrophication freshwater	kg P eq	0.00046507
Eutrophication marine	kg N eq	0.00603028
Eutrophication terrestrial	mol N eq	0.04573357
Ecotoxicity freshwater	CTUe	3.35975512
Land use	Pt	169.056448
Water scarcity	m3 depriv.	2.41958777
Resource use, energy carriers	MJ	139.424448
Resource use, mineral and metals	kg Sb eq	1.0077E-05
Climate change - fossil	kg CO2 eq	2.98252333
Climate change - biogenic	kg CO2 eq	0.00636037
Climate change - land use and transform.	kg CO2 eq	0.00170389

Table 6.17 Characterization Results: CED

Impact category	Unit	Total
Non renewable, fossil	MJ	47.4771453
Non-renewable, nuclear	MJ	95.7632488
Non-renewable, biomass	MJ	0.00278985
Renewable, biomass	MJ	15.7532479
Renewable, wind, solar, geothe	MJ	1.15017196
Renewable, water	MJ	4.4152042

Table 6.18 Characterization Results: ReCiPe 2016

Impact category	Unit	Total
Global warming (w/ C seq.)	kg CO2 eq	2.97750822
Stratospheric ozone depletion	kg CFC11 eq	3.1001E-06
Ionizing radiation	kBq Co-60 eq	4.97497229
Ozone formation, Human health	kg NOx eq	0.00922196
Fine particulate matter formation	kg PM2.5 eq	0.03125294
Ozone formation, Terrestrial ecosystems	kg NOx eq	0.0093916
Terrestrial acidification	kg SO2 eq	0.10477131
Freshwater eutrophication	kg P eq	0.0004654
Marine eutrophication	kg N eq	0.00074387
Terrestrial ecotoxicity	kg 1,4-DCB	11.3060982
Freshwater ecotoxicity	kg 1,4-DCB	0.09471725
Marine ecotoxicity	kg 1,4-DCB	0.12796669
Human carcinogenic toxicity	kg 1,4-DCB	0.09799022
Human non-carcinogenic toxicity	kg 1,4-DCB	1.77736001
Land use	m2a crop eq	0.75219349
Mineral resource scarcity	kg Cu eq	0.01170772
Fossil resource scarcity	kg oil eq	1.03844582
Water consumption	m3	0.06327468

Table 6.19 Characterization Results: ReCiPe 2008

Impact category	Unit	Total
Climate change	kg CO2 eq	2.93757574
Ozone depletion	kg CFC-11 eq	9.3026E-07
Terrestrial acidification	kg SO2 eq	0.10683697
Freshwater eutrophication	kg P eq	0.0004713
Marine eutrophication	kg N eq	0.00296371
Human toxicity	kg 1,4-DB eq	0.8056314
Photochemical oxidant formation	kg NMVOC	0.01902732
Particulate matter formation	kg PM10 eq	0.02385226
Terrestrial ecotoxicity	kg 1,4-DB eq	0.00115567
Freshwater ecotoxicity	kg 1,4-DB eq	0.04952682
Marine ecotoxicity	kg 1,4-DB eq	0.0456383
Ionising radiation	kBq U235 eq	4.82423243
Agricultural land occupation	m2a	1.54176552
Urban land occupation	m2a	0.13332779
Natural land transformation	m2	0.00048477
Water depletion	m3	0.0632171
Metal depletion	kg Fe eq	0.16588645
Fossil depletion	kg oil eq	1.04571738

Table 6.20 Characterization Results: CML

Impact category	Unit	Total
Abiotic depletion	kg Sb eq	1.0352E-05
Abiotic depletion (fossil fuels)	MJ	43.6485572
Global warming (GWP100a)	kg CO2 eq	2.94641917
Ozone layer depletion (ODP)	kg CFC-11 eq	9.2953E-07
Human toxicity	kg 1,4-DB eq	1.31177166
Fresh water aquatic ecotox.	kg 1,4-DB eq	0.90925612
Marine aquatic ecotoxicity	kg 1,4-DB eq	2055.23278
Terrestrial ecotoxicity	kg 1,4-DB eq	0.01219793
Photochemical oxidation	kg C2H4 eq	0.00515952
Acidification	kg SO2 eq	0.12590712
Eutrophication	kg PO4- eq	0.00410268

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