The Potential of Urban Agriculture in Combination with Organic Waste Valorization: Assessment of Resource Flows and Emissions for Two European Cities

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Glossary:

AD – Anaerobic digestion
BSF – Black soldier fly
CHP – Combined heat- and power units
DLI – daily light integral
FW – Food waste
GHG – Greenhouse gases
LCA – life cycle assessment
PAR – photosynthetically active radiation
UA – Urban Agriculture

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Abstract

Large scale urban agriculture (UA) has been considered as a potential contributor to the sustainability and the resilience of the food system. However, its synergy with the tackling of another main challenge in cities, namely the management of organic wastes, has previously not been extensively explored. The aim of this work is to understand the potential of an integrated food-energy-water-waste system in terms of food production, resource circularity and carbon footprint, in comparison with the current food and waste management systems in cities and an UA system without integration with waste management. Employing spatial analysis and a breadth of process parameters and city-specific data, the monthly material and energy flows were simulated and optimized for two European cities, Glasgow and Lyon. The results indicate a strong dependency of food self-sufficiency, waste assimilation and carbon footprint on the ratio between high-intensity greenhouse growing and urban gardening. For a chosen greenhouse to garden ratio of 1:4, the production of a large share of fruit, vegetable and fish supplied with nutrients from waste products and water from rainwater harvesting is theoretically possible, and the integrated UA systems could assimilate 51.7% and 16.9% of food waste in Glasgow and Lyon, respectively. From a food production perspective, co-located waste valorization could reduce the carbon footprint of the UA system by 7.8% for Glasgow and 12.6% for Lyon. However, overall emissions for the integrated UA system are 75.1% (Glasgow) and 4.3% (Lyon) higher than the current system, primarily due to high heating and lighting requirements for greenhouses in winter, even more so in Glasgow which is colder and with a carbon-intensive electricity grid. However, if a UA system is already in place, the decentralized valorization in combination with UA could provide significant emission reduction benefits compared to existing waste-to-energy schemes, in particular for insect rearing in combination with aquaponics. The outcome may inspire urban policy makers and stakeholders to consider the propagation of urban agriculture in symbiosis with decentralized manage waste strategies, in particular in areas with a low-carbon electricity grid, as a move towards self-sustaining cities and a more circular economy overall.

1 Introduction

Pressures on the current food supply chain in connection with climate change, increasing population and socio-environmental concerns call for alternative modes of food production and consumption. As a potential solution with multifold socio-economic implications, food production within urban environments has attracted considerable interests (Koegler et al., 2017). The productivity of large-scale urban agriculture (UA) for a city is determined by its potentially available area, which in recent years has been studied via geographic information systems (GIS) in 2D and 3D for a range of cities in the global North (Haberman et al., 2014; Saha & Eckelman, 2017; Sioen et al., 2018). Additionally, the yield per unit area is a decisive factor and differs depending on the growing practices, which range from open-air community gardening, over soil-based rooftop gardens to high-tech hydroponic and aquaponic greenhouses (Goldstein et al., 2016; Love et al., 2015). The environmental sustainability of these practices differs (Goldstein et al., 2016) and geographical location and the associated climatic factors play a decisive role (Benis et al., 2017). Building integration (Sanjuan-Delmás et al., 2018) and new distribution models (Kulak et al., 2013) provide a first step towards increased sustainability, but there is a strong need to further lower the footprint of this alternative food supply.

The main inputs for urban agriculture are plant nutrients (fertilizer), water and energy in form of heating and electricity, allowing for year-round production. Earlier work has mentioned the potentially positive role of UA in a city's metabolism or sensible use of resources (Deelstra, 1987). More recently, it has been suggested that specific urban waste streams can be synergistically used within UA (Mohareb et al., 2017) and urban nutrient flows currently considered as waste could satisfy fertilizing needs of urban farming endeavors (Wielemaker et al., 2018). Thus, to preserve resources and improve nutrient circularity, urban agriculture can be combined with bespoke organic waste management, making use of source separation and conversion technologies. Novel urban food and waste systems could be created that exchange nutrients, energy and other resources, while making use of a decentralized structure.
The urban utilization of products derived from organic waste is promising, such as nutrient-rich digestate in organic hydroponics (Shinohara et al. 2011, Kawamura-Aoyama et al. 2014, Rayns 2015), compost and digestate fiber in soil-based growing (Baltzell 2011, Grard et al. 2015, López-Mosquera et al. 2011) and insect protein production from organic wastes (Joly 2018, Salomone et al. 2017). The technologies for organic waste conversion, such as anaerobic digestion (AD), are well understood in isolation (Banks et al. 2011, Chiew et al. 2015) and are more recently integrated within a productive greenhouse (Zhang et al., 2013, Walker et al. 2017, Stoknes et al. 2016), allowing local heat utilization and production of fertilizer.

Despite the promising findings, there is currently a lack of detailed studies assessing the potential material and energy flows and associated environmental impacts at the city level of such novel urban food systems, which combine different growing practices with waste conversion, nutrient cycling and energy integration (e.g. provision of heat and upgrading of biogas). It also remains unclear how the choice between low-intensity gardening and high-intensity greenhouse growing affects the environmental performance of such systems (Weidner et al., 2019). Urban planning, investment decisions and resource allocation by decision makers and stakeholders could be enhanced by an understanding of what a city can expect if such systems are scaled up.

This work takes a quantitative modelling approach, synthesizing isolated literature information to investigate theoretical resource flows of a novel type of integrated UA & decentralized waste management system. It partially incorporates elements of urban metabolism methods, such as material flow and environmental footprint analysis with defined input and outputs, scope and boundary of the urban system under investigation (Barles, 2010). To assess how decentralized waste valorization can be symbiotically designed to cover resource needs of UA, this work undertakes the necessary step of determining the monthly production of food and by-products and the resulting resource requirements. Refined seasonal and crop granularity is employed to provide a more realistic description of the system. As the productivity depends strongly on the intensity of the growing practices, a variable ratio of growing intensities is investigated. Using the cities of Lyon, France, and Glasgow, UK, as case studies, the purpose of this work therefore is:

- To compare the influence of space and population characteristics of typical European cities and the intensity of the growing practices on the potential monthly self-sufficiency obtained for a selection of typical UA crops and fish;
- To assess the water, nutrient, electricity and heating requirements for the UA activities and how they are influenced by a different climate;
- To understand how the organic waste flows can best be utilized to support the UA activities from a nutrient and energy perspective; and
- To compare the operational greenhouse-gas (GHG) emissions of the current food system with the proposed integrated UA and waste management system on a monthly basis.

2 Methodology

2.1 Overview

This study evaluates the productivity (i.e. self-sufficiency), resource requirements and environmental performance of an integrated UA & decentralized waste management system via mathematical modelling and optimization of monthly material and energy balances. The overall methodology is illustrated in Figure 1. For the benefit of comparing different urban space and population characteristics, the dense urban City of Lyon (i.e. the 9 arrondissements) and the more spread out Glasgow City Council area were chosen (climate and population characteristics are listed in Table 2 in the result section). Glasgow has ongoing urban agriculture projects

(White & Bunn, 2017) and Lyon is a pilot city for novel decentralized organic waste valorization in UA (www.decisive2020.eu/lyon). To allow for a reliable assessment of the potentials of waste integration, a robust approach on the productivity and resource requirements of UA was followed. Two main tasks were therefore performed, the simulation of the growing process and its related resource requirements, and the optimization of the waste management aimed at minimizing the carbon footprint. The first task includes determining the available area (described in detail in SI A.1), assessing season-dependent food production and fertigation needs, and climate-dependent energy and water requirements. Of those, the determination of available growing area includes the allocation of areas to specific growing practices with differing resource intensities (illustrated in Figure 2). This allocation was represented by ratios between the practices (see Table 1). The second task includes defining the possible waste management pathways (referred to as the resource nexus), quantifying carbon footprint of all inputs and operations within the system considered and optimizing the allocation of waste streams between possible destinations.

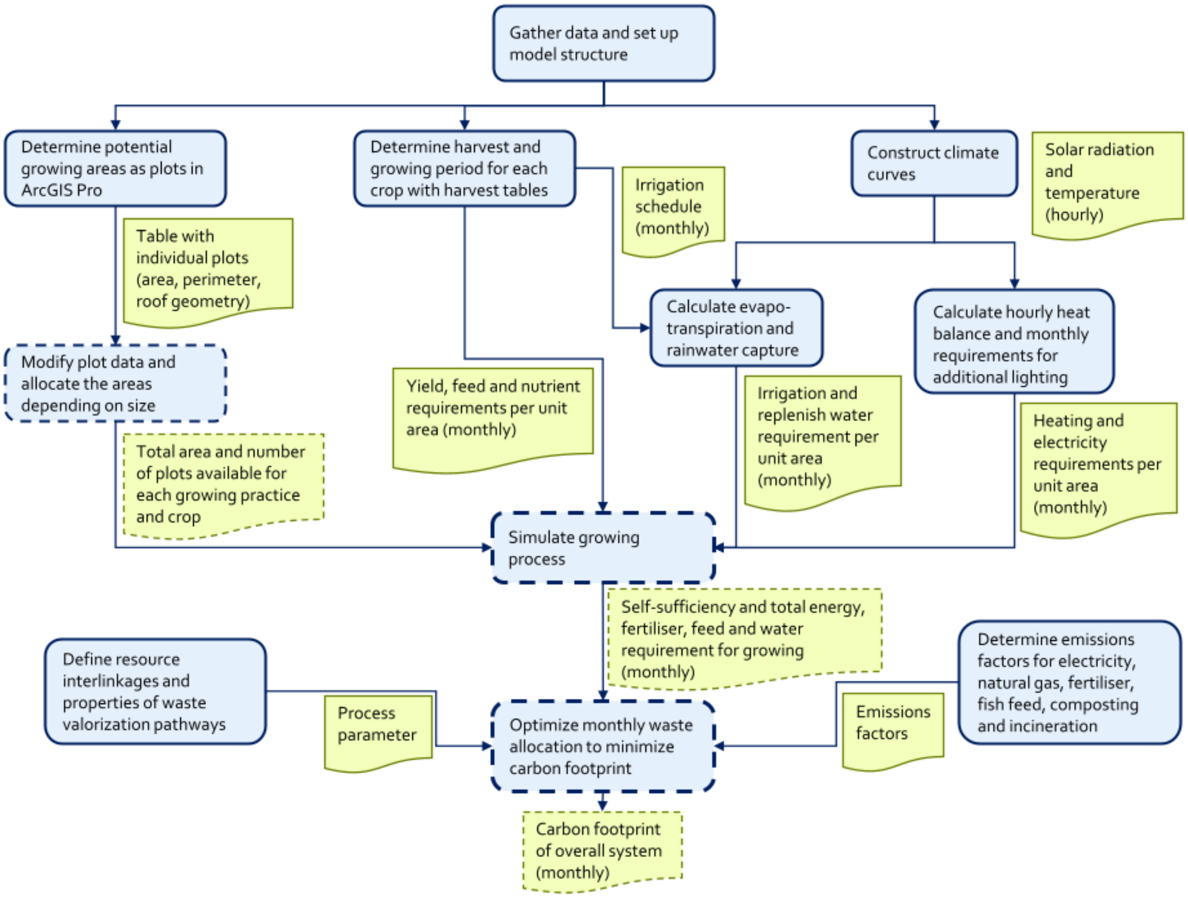

Figure 1: Overall methodology, the steps in dotted lines were repeated for different levels of growing intensity

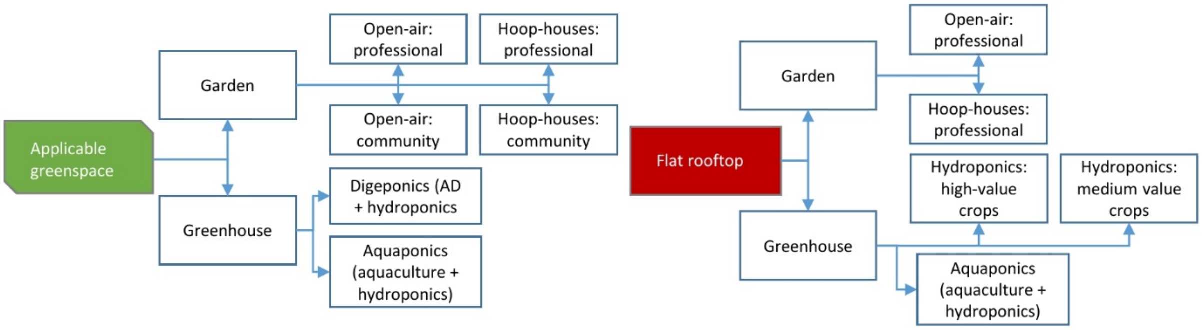

Figure 2: Different growing practices employed on greenspaces and rooftops

Table 1: Area use and ratios applied in spatial analysis

	Maximum area use or ratio	**Details**
Public greenspaces	50%	Maximum realistic or desired amount of conversion
Private yardspaces	25%	Maximum realistic or desired amount of conversion
Flat rooftops	20%	Adjusted considering structural integrity and access with minimal retrofit (Nadal et al., 2017)
Open-air vs hoop-houses	50/50	Equal share, hoop-houses extend season but open-air gardens could offer more urban ecosystem services
Professional vs community gardens	50/50	Equal share, focus either on productivity (higher yield, automated drip irrigation) or social benefits
Greenhouses versus gardens, on large ground plots (>750m^2)	Variable, detailed study on 20/80	Affected by preferred level of resource intensity (with greenhouses being more intensive), and public acceptance (e.g. aesthetic reasons)
Aquaponic vs digeponic greenhouses, on ground	50/50	Both equally desired (Alshrouf, 2017; Bowman, 2017; Pantanella et al., 2012)
Gardens vs greenhouses, on large rooftops	25/75	Assumption that on 25% of applicable flat roofs greenhouses are not allowed (e.g. fire safety) or desired, else greenhouses are preferred on rooftops
Aquaponic vs hydroponic greenhouses, on rooftops	25/75	Assumption that only 25% of applicable flat roofs have required structural integrity

The resource nexus that encompasses the considered material and energy connections between waste processing, resources extraction and growing is shown in Figure 3. The organic waste (brown) generated from different sources (dark grey) is converted via the decentralized treatment pathways (red) to inputs for the UA practices (light green), which in turn produce food (dark green). Surplus products from waste treatment have to undergo secondary treatment (brown-lilac) before final disposal. The term digeponic is used to describe hydroponics fertililzed by digestate by-products (Stoknes et al., 2016) but in this study the term hydroponics will be used irrespective of the origin of fertilizer. Details on the waste generation and processing are given in section 2.4. Details on system boundaries, i.e. which flows were considered to calculate resource requirements and carbon footprints, are described in sections 2.3 to 2.5 and visualized in the SI (F.5).

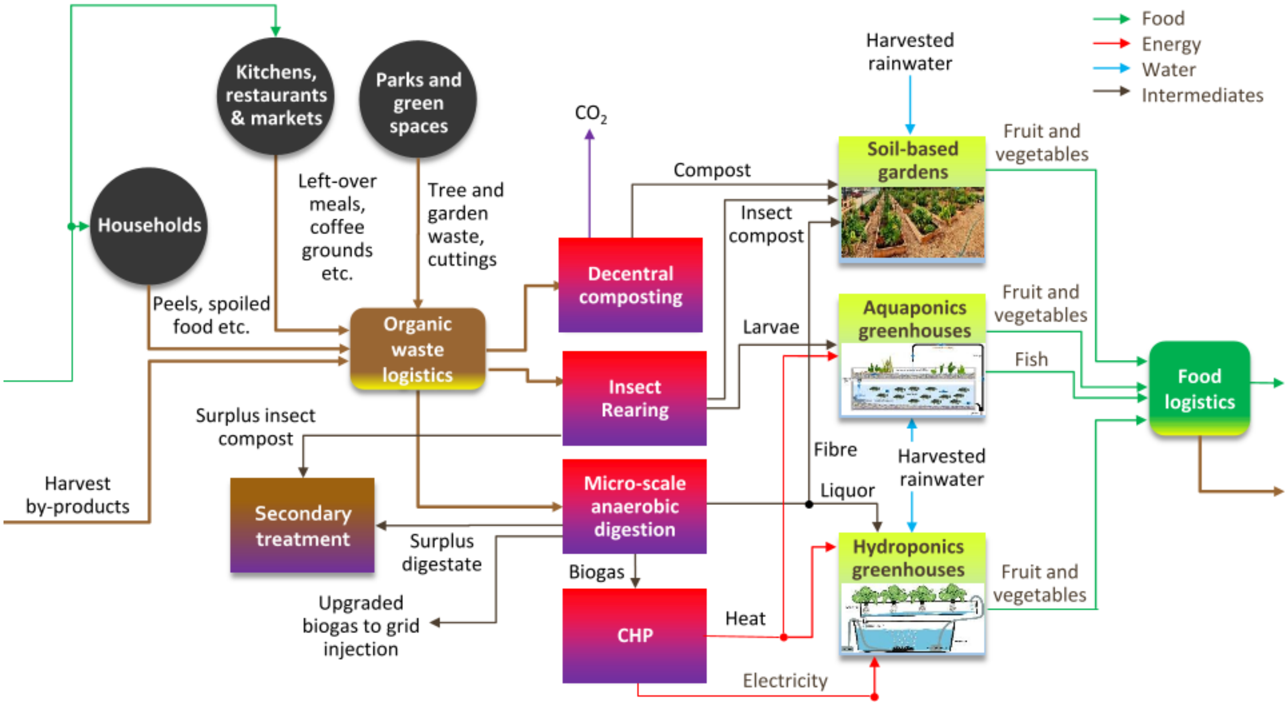

Figure 3: Resource nexus of integrated UA and waste management system

2.2 Food demand and production

The assessment of resource flows and emissions intended by this study needed to build on the selection of appropriate crops for UA, their demand profile and yield figures, the growing and harvest periods, as well as fertilization requirements. Crops were selected based on their applicability for UA in the Northern hemisphere as discussed in other studies (Clinton et al., 2018; Colasanti & Hamm, 2010), including specifically 6 fruits and 14 vegetables which constitute a significant share by weight of the fruit and vegetables consumed in the EU. To allow a comparison of self-sufficiency between the two cities, the recommended amount of 500 g/day/person of fruit and vegetable (Haberman et al., 2014) was split by 1:2 between fruit and vegetable (Health Canada, n.d.) and excluded exotic fruits (e.g. banana and pineapple, considered to be 50% of fruit demand), resulting in 433 g/day/person. This amount was then distributed over the 20 crops according to actual consumption patterns in the EU. Yield figures for soil-based growing were determined by adjusting FAOSTAT yield data from 2016 (www.fao.org/faostat) by a factor of 0.75 for community garden and of 1.25 for professional gardening, which served as a feasible average of other studies with similar classification (Grewal & Grewal, 2012; Haberman et al., 2014; Sioen, et al., 2017), and 5% loss through refuse was assumed (which is referred to as "plant biomass" in this study). Hydro- and aquaponic yields for eight applicable crops were employed from case studies mentioned in Haberman et al. (2014).

As one of the aims of this work is to quantify the potential monthly productivity of UAs, the annual yields for open-air gardens and hoop-houses were split up on a monthly basis according to a harvest table, which included potential growing season extension in hoop-houses for 9 crops and potential storage of 5 crops (Colasanti & Hamm, 2010). Together with crop growing cycles, the harvest tables were used to determine the additional productivity of crops that benefit from increased yield through season extension in hoop-houses (more information in SI, B.7-9). It was assumed that the 9 crops which benefited from the growing season extension would be grown in equal ratios in hoop-houses while the remaining 11 crops would be grown in equal ratios in open-air gardens. The allocation of crops within year-round high-tech greenhouse growing was determined such that the increase in self-sufficiency due to greenhouse growing is equal between different crops (method see SI, A.10). All related figures and their sources can be found in the SI (A.10 to B.08).

Aquaponic greenhouses produced not only the 8 hydroponic crops but also fish including carp, catfish and tilapia (detailed parameter in E.3 in SI). The available area was divided by the ratio in consumption of those fish (Fisch-Informationszentrum, 2016).

2.3 Operational requirements for growing

Energy was used by the system primarily as electricity (for artificial lighting, irrigation pumps and other equipment) and as heat for greenhouse climate control, insect rearing and AD operations (see section 2.4). Assessing the monthly heating and additional lighting requirements of greenhouses required solar radiation and ambient temperature profiles, which were created from 30-year average data (method described in SI, Section C). The electricity requirements for artificial lighting were then calculated according to the difference between plant-specific photosynthetically active radiation (PAR) needs as daily light integral (DLI, in SI B.5) (Shao etl al., 2016) and the naturally provided PAR, incorporating molar efficacy of the lighting system and shading and transmittance factors (see D.2 in SI). Heating requirements in greenhouses were calculated on an hourly basis employing the constructed climate curves, fixed day- and night-minimum temperatures and relations for five predominant fluxes while omitting any thermal inertia from the structure. Furthermore, reduced heating requirements for the host buildings in case of rooftop greenhouses were considered similar to Goldstein et al. (2017), hence the typical user heating requirements for France and the UK were assumed to be reduced by 3%. All equations and parameters are presented in the SI (D.1 and D.2).

Irrigation and replenish water requirements were calculated for both, open-air and hydroponic growing (C.6 in SI). Rainwater harvested in the integrated scenario on hoop-houses and greenhouses, used for irrigation and to supply AD feed water, was investigated considering three different scenarios:

a) No rainwater storage
b) Short-term local storage, i.e. use within month in same facility only
c) Shared reservoir, i.e. distribution network with intra-monthly storage capacities

To assess nutrient supply, nitrogen, phosphorus and potassium (NPK) were considered individually as essential plant nutrients. It was assumed that aquaponics farms can satisfy their nutrient needs internally while hydroponically grown plants rely on added nutrients recirculating in solutions, determined by their nutrient uptake of the (edible) crops (B.9 in SI) and the nutrient uptake of non-edible plant material (considered to be 25% of the crop weight, e.g. leaves and stems), also accounting for some recovery loss (Bugbee, 2004). The fertigation of soil-based growing was according to FAOSTAT fertilizer rates for France and the United Kingdom (B.10 in SI).

2.4 Waste generation and processing

It was assumed that both, households and commercial sources generate 0.2 kg/day/capita of source-separated food waste each (Mugica & Spacht, 2017; Wielemaker et al., 2018) while yard waste generation depended on season (David, 2013) and non-built up area (1 kg/m^2/year) (Adwiraah et al., 2012). It is important to note that the generated food waste stems from the supply and consumption of all food products, not just UA produce. Relevant composition parameters for both types of waste, including NPK content, can be found in the SI (E.1 and E.2). It was assumed that refuse generated as harvest by-product has the same composition as yard waste and they together constitute the generated green waste (GW).

As the most traditional conversion technology, aerobic decomposition via small decentralized composting units was considered in this study to convert organic waste into soil conditioner that provides nutrient for open-air growing. Co-located near greenhouses, insect rearing of black soldier fly (BSF) fed on food wastes was employed to supply protein for the fish raised in aquaponic greenhouses. It was assumed that the dry mass of live BSF larvae can replace 25% of conventional fish meal without fish growth reduction (Salomone et al. 2017). Mesophilic wet anaerobic digestion via small- and micro-scale digesters co-located with greenhouses were employed to convert various organic wastes into digestate and biogas, which is either upgraded and fed to the grid or burned in a combined heat and power Stirling unit (CHP) if there is sufficient heat demand in greenhouses. The digestate was locally separated into liquid (liquor) and solid (fiber) using a small-scale centrifuge and the liquor made available as fertilizer for hydroponic growing, also called digeponic (Stoknes et al., 2016), and the fiber as fertilizer for soil-based growing. Any surplus digestate products had to be treated semi-centrally for final disposal and the final concentrate was sent to incineration together with surplus insect compost. For all three main conversion technologies, a processing time of 1 month was assumed, hence changes in feed composition and quantity had an effect on the products only in the following month. Further, it was assumed that waste products are converted in close proximity to UA operations and thus transport impact was omitted. All other considerations and parameters are listed in the SI (E.3 and E.4).

2.5 Description of scenarios and operational emissions

To compare the benefit of integrating decentralized waste valorization, three different scenarios were carried out:

- The integrated scenario (referred to as "integrated"), in which scaled-up urban growing operations may be supplied by feed and fertilizer from converted waste products and benefit from co-locating CHPs for heat integration and electricity generation. The material and energy flows were determined via optimization as described in Section 2.6.
- A scaled-up UA scenario but without integration or decentralized waste valorization (referred to as "UA+Ext"). All inputs to UA were externally supplied (fish feed, chemical fertilizer, natural gas, grid electricity) and all waste streams externally (i.e. outside the city boundaries and the UA system) treated (incineration with energy recovery and windrow composting)
- The current system (referred to as "current"), in which food is supplied from outside of city boundaries (as described below) and waste treated externally as in the "UA+Ext" scenario

Emissions and resource use were compared on an operations-only basis (compared to the full life-cycle), hence do not include impacts from facility construction and other equipment and materials required for setting up and running the system (for implications see Section 4.4). To assess the environmental benefits of the “integrated” UA scenario versus the operations of the “current” scenario, it was assumed that the produce would replace either imported or nationally produced food. The share of supply that was considered imported was determined using production, export and import data from FAOSTAT and then distributed over 12 months in the year according to the harvest table and other considerations (e.g. year-round greenhouse production of tomatoes and cucumber in the UK) (B.3 and B.4 in the SI). The carbon footprint of national and imported food supply for the UK was based on Audsley et al. (2009), which estimated the associated emissions per crop for production and transport until the regional distribution centers. The carbon footprint of national and imported food supply for Lyon was extracted from a range of LCI databases (farm-gate) and adjusted for transport emissions similar to the UK values (described in F1 in SI). The carbon footprint of externally supplied food thus comprises all the emissions during the entire process of food production and supply, including those linked to energy and water use at various stages. The carbon footprint of the produce distribution within the city was then assumed to be the same for the “integrated”, “UA+Ext” and “current” scenarios and thus excluded in the assessment. Fish type specific production and distribution emissions were used for the “current” scenario. Insect meal would replace the share of equivalent soy meal as part of the overall accounted feed in the “integrated” scenario. The associated emissions for the production (for both soil-based and hydroponic growing) and application of externally supplied chemical fertilizer (for soil-based growing only) were taken from EU guidelines. The carbon footprint of grid electricity was accounted with 0.06 and 0.41 kg CO_2e/kWh for France and UK respectively, natural gas heating with 0.27 and 0.20 kg CO_2e/kWh for France and UK respectively and water with 0.15 kg CO_2e/m^3. The low carbon footprint of electricity in France is due to nuclear energy provision, and can be viewed as a representative of a low carbon electricity grid, such as one with a high-level penetration of renewables. All carbon footprints and emissions factors, including their sources, are listed in the SI (F.1-F.4).

In the “current” scenario, it was assumed that food waste is not source-separated and is thus incinerated together with municipal solid waste (waste-to-energy) while green waste is composted in windrows, both outside of the city boundaries (20 km distance, diesel truck). This work does not consider the final treatment of incineration ash but takes into account the energy recovery in incineration facilities. The end use of compost from windrows was not considered as (i) the majority was considered not to be used in commercial food production and (ii) the benefits as fertilizer replacement were found to be negligible compared to the process emissions. The emissions during incineration were dependent on the (biogenic) carbon content and the total mass of the processed material. Gaseous losses during the composting process constituted the emissions for both decentralized and windrow composting. For the “integrated” scenario, any concentrate of surplus waste products processed semi-decentrally were also treated outside the city (co-incineration). The emissions arising from treating the fraction of the primary waste (i.e. food and green waste) that was not used in the “integrated” scenario was not included in the calculation for any of the three scenarios, because this part of the emissions would be identical for all the scenarios and therefore not necessary to include for comparison. The applied (insect) compost and digestate fiber in the "integrated" scenario released 0.5% and 0.75% of available N as N_2O, respectively, and accounted for as well were the emissions from biogas burning, energy requirements for secondary treatment of surplus waste products, and the credits of biomethane (grid injection) for the replacement of natural gas (production only, excl. combustion).

2.6 Optimization of waste allocation

The sensible allocation of primary waste (i.e. food waste and green waste) to valorization technologies and how waste products are allocated to either urban agriculture or secondary treatment would be a primary concern for future integrated productive urban food systems. Among the options of reducing external material input (e.g. fertilizer and feed), overall energy requirements and the carbon footprint, the last one was taken as the overall sustainability indicator. Non-linear optimization was conducted to minimize the total GHG emission from all months as shown in the objective function (Eq. (1)). Monthly GHG emissions from external heating (Q^{Ext}), electricity (P^{Ext}), NPK fertilizer ($Fert^{Ext}$) and fish feed ($Feed^{Ext}$) requirements are multiplied with the respective GHG footprints, (i.e. the GWP terms) and added to the GHG emissions (i.e. the GHG terms) resulting of biogas combustion (GHG^{Biogas}), fertilizer application (GHG^{Fert}), waste incineration ($GHG^{Incineration}$) and composting ($GHG^{Composting}$).

$$\text{Min} \quad GHG^{Total} = \sum_{m \in M} \begin{bmatrix} Q_m^{Ext} * GWP^{Gas} + P_m^{Ext} * GWP^{Power} + Fert_m^{Ext} \\ * GWP^{Fert} + Feed_m^{Ext} * GWP^{Feed} + GHG_m^{Biogas} \\ + GHG_m^{Fert} + GHG_m^{Incineration} + GHG_m^{Composting} \end{bmatrix} \qquad (1)$$

The decision variables were the amount of food and green waste sent to each valorization pathway and the amount of waste products assimilated by the UA operations. The constraints included:

- Over-fertilization was not allowed, meaning that when one of the plant growth nutrients was saturated through waste products while the other nutrients were still insufficient, the shortage would be met by corresponding externally supplied fertilizers;
- Composting to be an option only for existing demand in the following month;
- The feed into AD to consist of at least 50% food waste;
- The feed into composting to consist of at least 25% food waste;
- Insect rearing was chosen only up to the point where larvae constitute up to 25% of fish feed; and

- The CHP units to be running when and only when the heating requirements of the co-located ground-based greenhouses was no less than the potentially available heat extracted from CHP, in order to avoid undesirable loss or unrealistic derivation of surplus heat
- No secondary treatment of digestate liquor, i.e. all digestate liquor that is produced had to be used for hydroponic growing (but without overfertilization)

The introduction of the last constraint was due to the following considerations: From an initial assessment of the material flows, it was found that the optimization - without restricting the secondary treatment of streams arising from waste processing units - led to all waste being treated within city boundaries via either AD, composting and insects while generating a large surplus of digestate products (shown as Sankey diagrams in G.4 in SI). This extensive generation and decentralized treatment of surplus waste product is most likely not practical nor desired in urban areas, hence the additional constraint of no decentralized treatment of digestate liquor was introduced (more liquor than fiber was used in the system, surplus fiber was still allowed and underwent secondary treatment).

The detailed mathematical model, including the list of constraints, variables and parameters is described in the SI (G.1-G.3). Excel in combination with What'sBest!® was used for modelling and optimization.

3 Results

3.1 Spatial analysis and climate factors

Table 2 lists the main differences between the two case study locations with respect to urban layout, climate and the resulting energy demand by greenhouses. The urban layout influences the potential productivity and the possible split between year-round greenhouse growing and seasonal harvests in gardens. A graphical representation of the spatial analysis results is shown in H.1 in the SI.

Table 2: Relevant spatial and climatic factors for Glasgow and Lyon

Aspect	Sub-aspect	Glasgow	Lyon
Spatial factors	Total area (km^2)	186.86	47.09
	Population density (people/km^2)	3560	6819
	Share of flat rooftops of total rooftop area (%)	7.3	21.5
	Share of applicable rooftop area of total urban area (%)	0.8	4.7
	Share of applicable greenspace area of total urban area (%)	8.1	3.1
Climate factors	Average temperature (°C)	8.73 °C	11.59 °C
	Annual rainfall (mm)	1200	844
	Average daily light integral (mol PAR / m^2)	13.8	18.8
Greenhouse energy demand	Heating intensity (kWh/m^2/year)	353	237
	Lighting intensity (kWh/m^2/year)	252	143

3.2 Influence of intensity of growing practices

The resource intensity of soil-based gardening and high-intensity greenhouse growing – with heating and artificial lighting – is quite different. Thus the extent to which the latter is used on larger ground-based plots is a key variable in the assessment of the environmental performance. Figure 4 shows the carbon footprint of the optimized system for a varying ratio of greenhouses versus gardens on large ground plots (0%-100%), normalized for the "current" scenario. Also shown are the self-sufficiency achieved and waste assimilation (in relation to total waste and food waste only) for the varying ratio of greenhouses versus gardens. The waste assimilation is calculated on a mass basis and includes waste products used as fertilizer, fish feed or as biogas in CHPs but excludes biogas injected into the grid. It has to be noted that in the case of 100% ground-based greenhouses, soil-based garden growing will still be practiced on smaller plots and in the case of 0% ground-based greenhouses there will still be production in rooftop greenhouses. Furthermore, with an increasing share of high-intensity greenhouse growing, the productivity of the system rises, and the carbon footprint of the hypothetic operation of not only the "integrated" and "UA+Ext" but also in the "current" scenarios becomes greater as the consequence of producing the same elevated levels of output.

For both cities, supplying higher levels of self-sufficiency via greenhouses results in a higher carbon footprint than the current system. In both cities equal carbon footprints between the "integrated" and "current" scenario are only reached at lower percentages of greenhouse growing, i.e. 4.1% for Glasgow and 13.3% for Lyon. The assimilation of waste at high ratios of greenhouse growing approaches 50% in Glasgow due to the limitation on green waste as feed for AD.

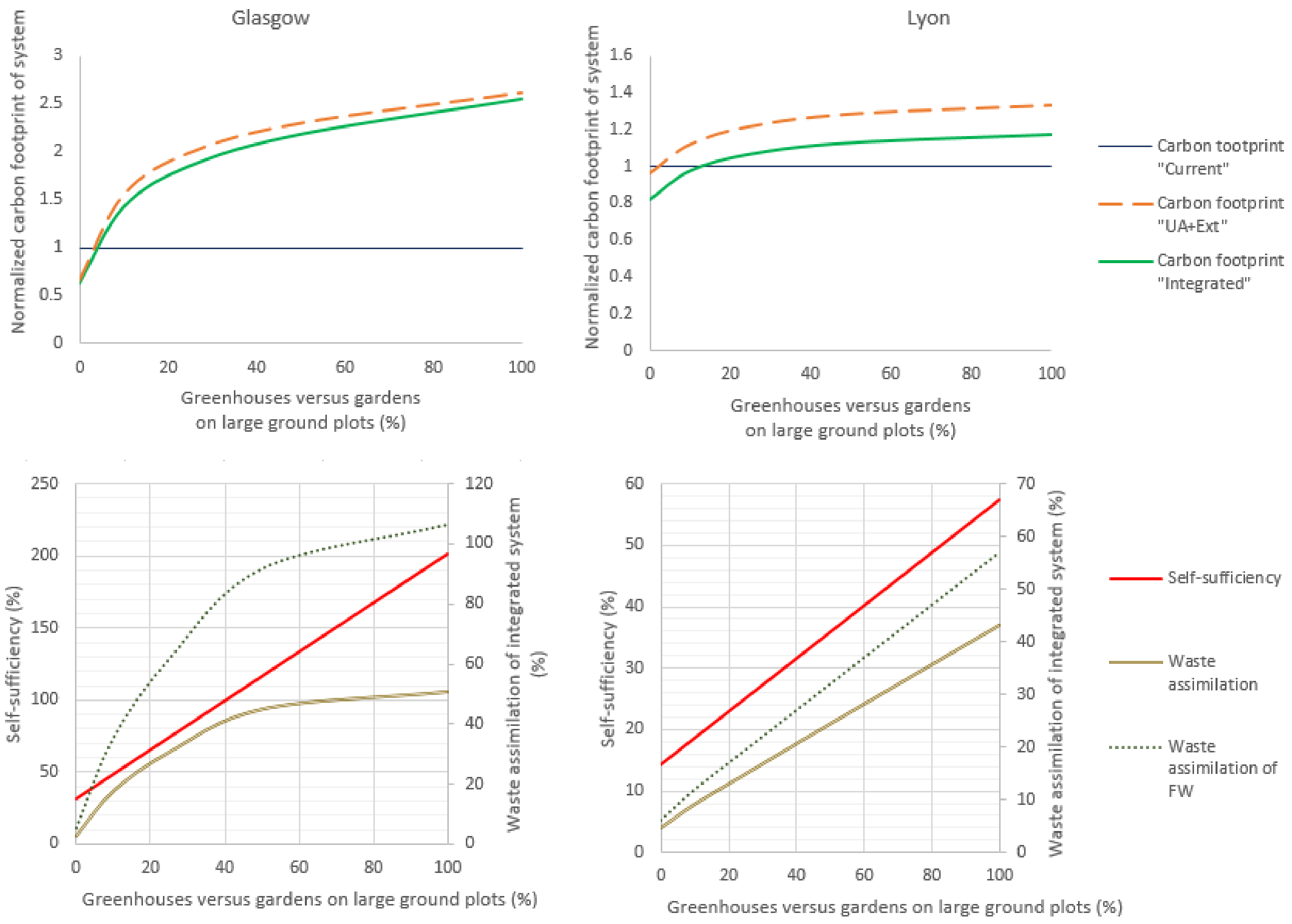

Figure 4: Impact of greenhouse to garden ratio.

Subsequent assessments and visualizations in this work that compare material and energy flows, monthly carbon footprints and integration benefits rely on a fixed ratio of greenhouses versus gardens for larger ground plots. As this work aims to also evaluate the influence of urban geography and climate, it is desirable to choose the same ratio (i.e. adoption level of greenhouse growing) for both cities, hence a few considerations have to be made. First, although a higher adoption level of greenhouses leads to higher self-sufficiency and waste assimilation in both cities, it also leads to a much higher carbon footprint than the “current” scenarios. Second, it is unrealistic to picture a scenario in which all applicable large urban greenspaces are converted to greenhouses, which are likely less (socially) inclusive and aesthetic than gardens. Third, for a resilient urban food system, it is likely advisable to supply a good proportion of, but not all, fruits and vegetables from within urban boundaries even during winter months. Thus, a compromise is assumed at 20% greenhouses and 80% gardens on large plots, resulting in a similar carbon footprint for at least one of the cities, Lyon, between the “integrated” and the “current” scenario while still providing a considerable level of winter crop production and assimilation of organic waste in both cities. The resulting distribution of applicable parcels to the growing practices is shown in H.2 in the SI.

3.3 Results for a 20% adoption level of ground-based greenhouses

3.3.1. Self-sufficiency

Figure 5 shows how the harvest and fish production each month satisfies the demand in each city. As expected, the seasonally limited production in open-air and hoop-house growing leads in both cities to a drop in production during the winter months. The self-sufficiency in winter can be seen as “baseline” supply from all-year operating greenhouses, indicating the maximum realistically achievable fresh food supply during those months. Combining fruit and vegetables, UA in Glasgow achieves in average 65.4% self-sufficiency and 22.9% in Lyon.

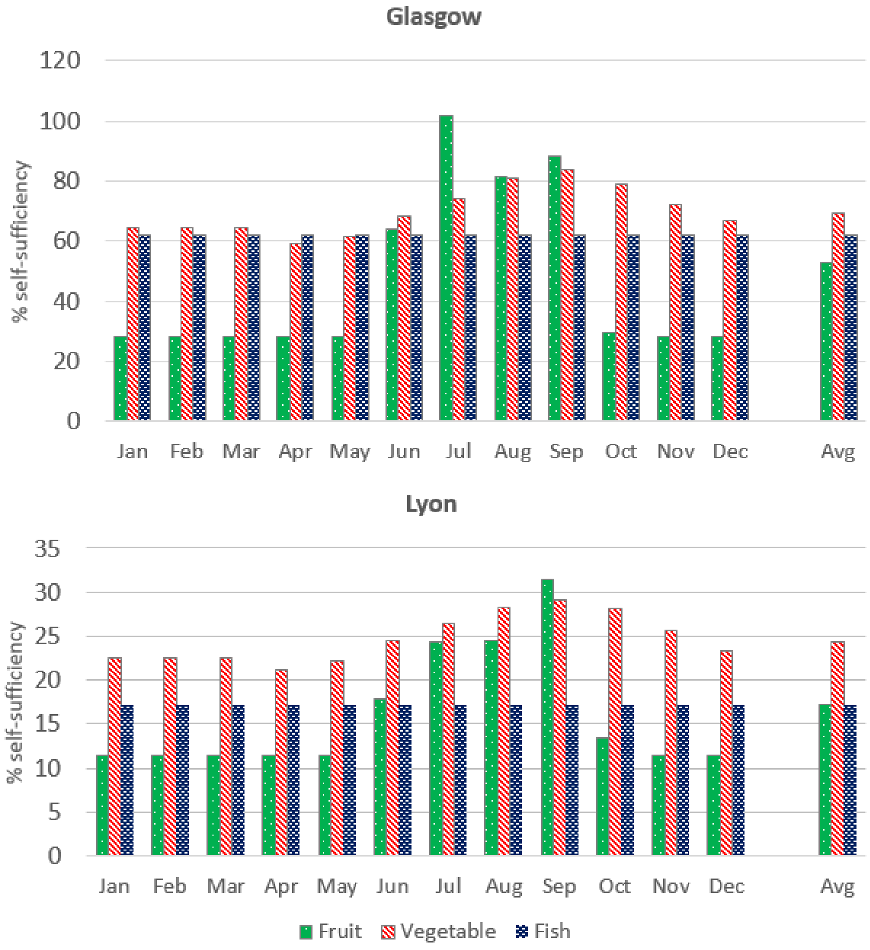

Figure 5: Annual self-sufficiency profiles for Glasgow and Lyon

3.3.2. Resource intensity

Table 3 shows the resource use and other metrics for both cities. The water use is split by the three rainwater harvesting scenarios for both soil-based and soil-less growing and only meets the irrigation and replenishing water requirements of the urban produce. The negative values for hydroponics in the shared reservoir scenario occur because more rainwater is harvested throughout the year than the annual requirements for hydroponic greenhouses. For Glasgow, the shared reservoir scenario covers all remaining water requirements within the whole system boundary (incl. water for AD and insect rearing operations).

The use of external fertilizer is higher overall in Glasgow due to a higher crop production while in both cities the "integrated" scenario still requires some external fertilizer due to the mismatch between the nutrient demand and the nutrient profile of the waste products, in particular for hydroponics.

Table 3: Resource metrics for Glasgow and Lyon

Aspect	Sub-aspect	Feature			
Water use (l/kg)	Soil-based gardening	No harvesting	Glasgow		**191.4**
			Lyon		**224.1**
"Integrated" scenario		Short-term storage	Glasgow		**108.9**
			Lyon		**144.7**
		Shared reservoir	Glasgow		**18.8**
			Lyon		**69.6**
	Hydroponics greenhouses	No harvesting	Glasgow		**23.3**
			Lyon		**28.3**
		Short-term storage	Glasgow		**1.1**
			Lyon		**5.3**
		Shared reservoir	Glasgow	-	**19.3**
			Lyon	-	**0.3**
Fertilizer use (tonne/y)	Gardening and hydroponics	"UA+Ext" scenario	Glasgow		**1774**
			Lyon		**322**
		"Integrated" scenario	Glasgow		**519**
			Lyon		**60**
Carbon intensity (kg CO2e/kg)	Tomatoes from greenhouses	"UA+Ext" scenario	Glasgow		**3.88**
			Lyon		**1.97**
		"Integrated" scenario	Glasgow		**3.83**
			Lyon		**1.93**
Waste integration (%)	Assimilation within UA-System	Total decentral utilisation	Glasgow		**25.8**
			Lyon		**13.0**
"Integrated" scenario		Considering food waste only	Glasgow		**51.7**
			Lyon		**16.9**
	CHP operation	CHP-utilisation	Glasgow		**68.7**
			Lyon		**53.5**
	Resource recovery	P-recovery	Glasgow		**13.5**
			Lyon		**5.5**

The carbon intensity of growing tomatoes in greenhouses may serve as comparison between the cities and with other studies, which are discussed later in the paper. It considers all electricity and heating requirements for the "UA-Ext" scenario and in addition the effect of the local heat utilization from the CHP and that of the use of digestate liquor fertilizer for the "integrated" scenario but excludes credits from biogas injection. Reductions are with 1.13% for Glasgow and 1.99% for Lyon rather small.

The waste integration parameters describe the (nutrient) circularity factors of the "integrated" scenario. In both cities, the assimilation of the primary waste, in the form of products derived from waste processing that are consumed within the system (i.e. excluding streams undergoing secondary treatment and the bio-methane for grid injection), is far from complete, meaning that only a small share of the generated organic waste may be utilized by a considerably scaled-up UA system. Considering food waste only, the potential assimilation would increase and in the city of Glasgow reach around 50%. The CHP utilization, i.e. the percentage of time the CHPs are operated within the year, is constrained by heating requirement rather than biogas quantity. The large share of non-assimilated waste correlates with the low recovery of phosphorus as shown in table 3.

Concerning the feasibility of allocating urban space for waste conversion, the total area required for insect growing in Glasgow and Lyon was 177,813 m^2 and 24,416 m^2, respectively (0.095% and 0.050% of total city area), and the total digester volume required to convert the waste in Glasgow and Lyon was 4951.1 m^3 and 1315.4 m^3, respectively.

3.3.3. Waste allocation

For both cities, material flow diagrams are presented which show the waste valorization pathways and the annual mass flow in kilo tonnes between them, determined by solving the optimization problem (Figure 6). In those diagrams, each pair of text (e.g. "Food waste") and number (e.g. "93.086") relates to the bar to its left for the first two stages and to the bar to its right for the last two stages. The products of incineration and windrow composting have no final use in the UA system and are thus not continuously displayed beyond the first panel. For Glasgow, the AD takes in the maximum permissible amount of green waste, despite the resulting lower biogas yield. This is because for Glasgow, the emissions from windrow composting of green waste (0.443 kg CO_2e/kg green waste, same as Lyon) is higher than those from incineration of food waste (0.414 kg CO_2e/kg green waste, slightly lower than that for Lyon). In both cities, the primary waste that is assimilated goes to AD and insect rearing and none to decentralized composting as chosen by the optimization based on lowest emissions. Hence, insect compost and digestate fiber are chosen for fertigation of soil-based UA. Due to the different nutrient profiles, combining the two sources allowed maximization of the total amount of organic fertilizer applied and hence minimization of the secondary treatment emissions. On the other hand, production of digestate fiber and insect compost in both cities is about twice as high as the use potential, making half of it an unwanted by-product to be treated. The prevention of overfertilization with a single nutrient was in both cities the limiting factor for further utilization of organic fertilizer; in soil-based growing N had to be externally supplied and in hydroponics P and K.

3.3.4. Carbon footprint comparison

Figure 7 displays the monthly emissions for the “integrated” and “current” scenarios for both cities, split by heating, electricity (excl. secondary waste treatment), other inputs (fish feed, external fertilizer and water), and all emissions related to waste management (incl. production of organic fertilizer and secondary waste treatment). The total carbon footprint is strongly influenced by the carbon intensity of the electricity grid, which is much higher in Glasgow. The figures show significantly increased emissions for the “integrated” scenario in the winter months, mostly driven by artificial lighting needs and heating requirements. For both cities, the “current” scenario has a significantly greater footprint in summer. This is partly because the F&V production accounted for within the “current” scenario was assumed to match the higher productivity of the UA-system in summer. It has to be noted that the share of imported food with a larger carbon footprint is higher in Glasgow than in Lyon, resulting in greater average emissions (1.39 and 1.23 kg CO_2eq/kg produce for Glasgow and Lyon, respectively). Comparing the average carbon footprints, the “integrated” scenario emits significantly more CO_2e than the “current” scenario in Glasgow (by 75.1%) and slightly more than the “current” scenario in Lyon (by 4.3%).

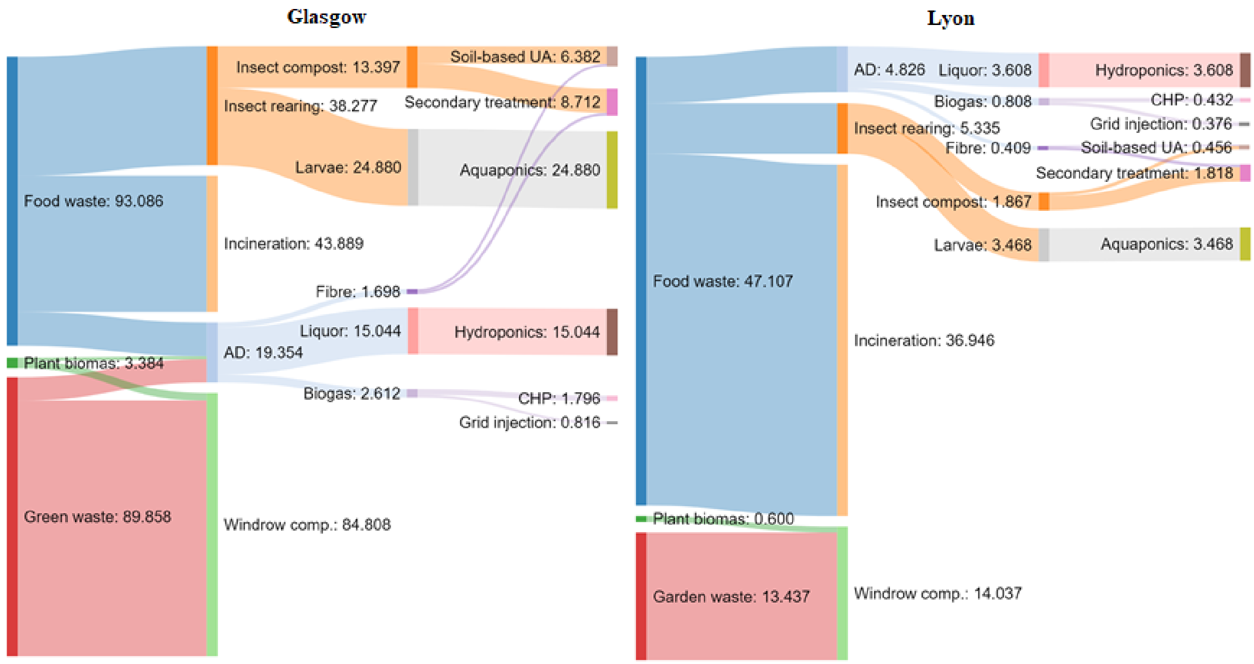

Figure 6: Sankey diagram for waste flows in Glasgow and Lyon (kt/year)

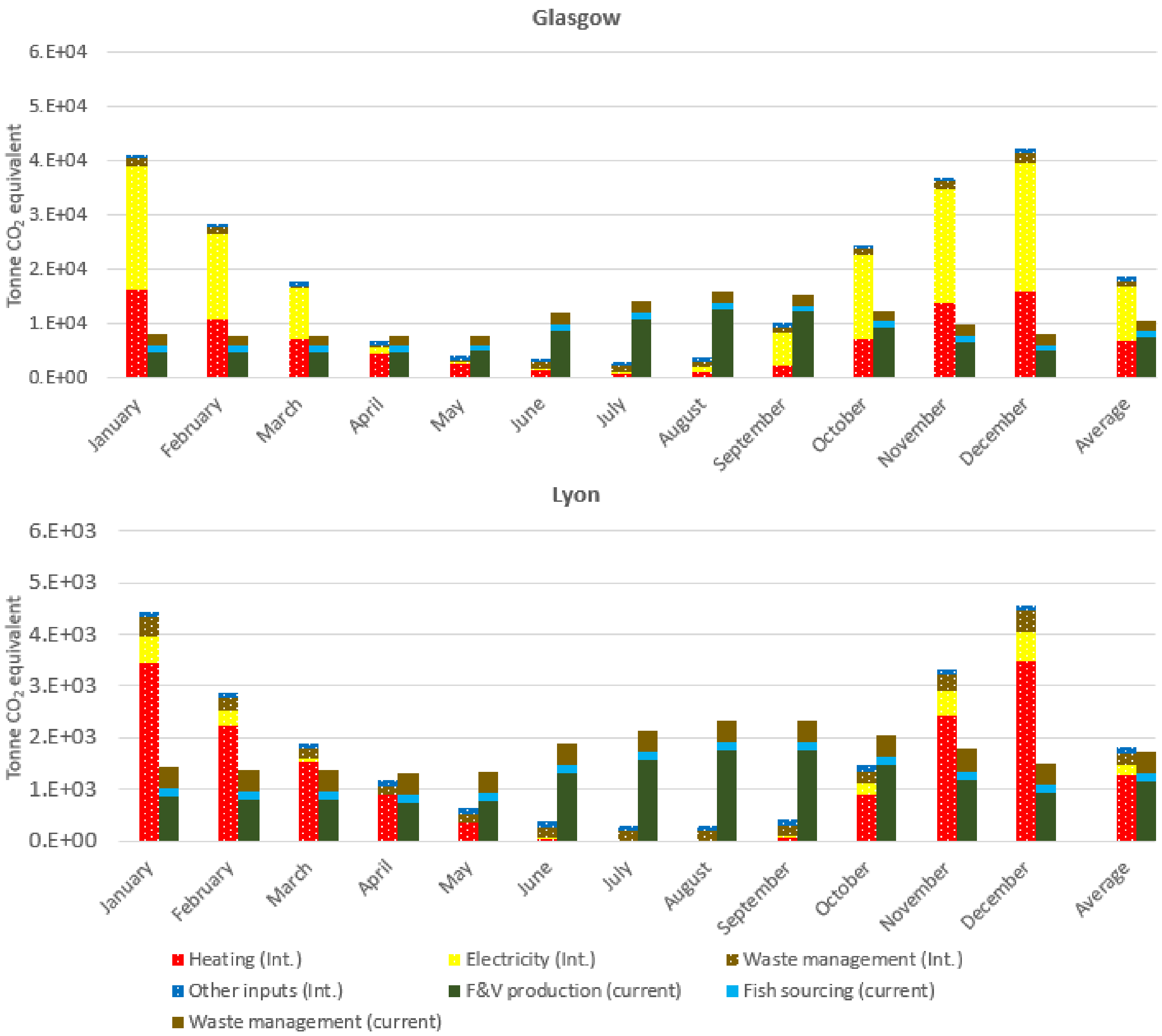

Figure 7: Annual emission profile of the "integrated" and "current" scenarios for Glasgow and Lyon

3.3.5. Benefit of integrating UA with waste management

Figure 8 shows the difference in annual emissions between the "UA+Ext" scenario without integrated waste valorization and the scenario of UA combined with decentralized organic waste valorization for both cities. To compare the two cities despite their different production levels, the emissions are displayed per kg of produce.

Due to the higher productivity relative to the waste assimilation (which is the waste accounted for in the "UA+Ext" scenario) and the lower associated emissions for incineration in Glasgow, the per-kg-produce emissions for waste are lower than in Lyon.

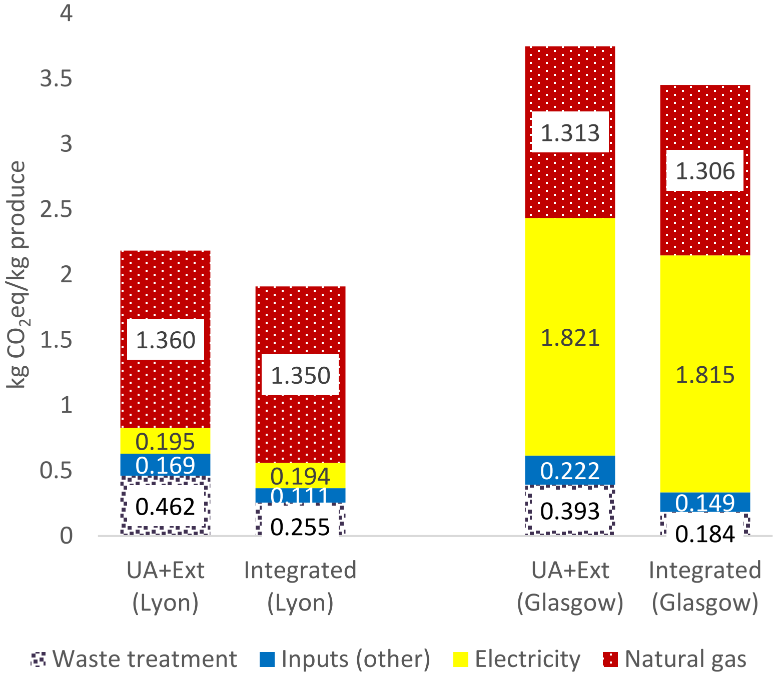

Figure 8: Emission intensity of the "UA+Ext" and "integrated" system for Glasgow and Lyon

Despite the production of electricity and heat by converting biogas in a CHP, the "integrated" scenario has a rather insignificant reduction in emissions from grid electricity and gas use compared to the "UA+Ext" scenario with external waste treatment. This is mainly due to (i) the parasitic electricity requirements for the operation of the AD and the centrifugal solid/liquid separation and (ii) the high heating requirements in relation to the heat produced.

In both cities, the emissions associated with waste treatment receive significant reductions in the "integrated" scenario. Further, compared with the lowering of the carbon footprint of energy inputs enabled by the integration between UA and waste management, the replacement of chemical fertilizer with waste products and especially of fish feed with insect larvae had the a considerably larger effect (included in the "Inputs (other)"). In total, the reduction of emissions (per kg of produce) due to the integration between UA and waste management was 7.9% for Glasgow and 12.6% for Lyon.

So far, the carbon footprint has been assessed from a UA perspective, where emissions are determined by the resource demands of urban food production and how these demands are met; the latter is particularly affected by the partial supply of inputs derived from organic waste products (electricity, heat, fertilizer, feed). This assessment has shown an apparent mismatch between (1) the high amount of digestate provided relative to the fertilizer requirements of UA and (2) the low amount of energy provided by AD relative to the energy requirements of UA. This mismatch leads to a low level of waste assimilation at the city level and reliance on further external energy inputs. From a waste management perspective, the potential benefits of waste valorization within a UA system (which may have been established for a much broader range of benefits) can be examined by assuming full assimilation of all waste products. Table 4 lists a few carbon footprint metrics for the treatment of 1 kg of food waste. The carbon balance compares these replacement benefits with the emissions for process inputs (e.g. natural gas, electricity) and biogas combustion; a negative sign means the replacement benefits are higher than the process emissions. The carbon balance for the integrated system is influenced by the ratio of primary waste sent to the more favorable insect rearing route (insect rearing benefits: -0.136 and -0.140 kg CO_2e/kg FW for Glasgow and Lyon, respectively, versus AD with 0.154 kg and 0.168 kg CO_2e/kg food waste for Glasgow and Lyon, respectively). In both cities, using the products from insect rearing and AD locally in urban agriculture provides significant benefit versus incineration with energy recovery. Theoretically, the benefits for AD could also hold true for centralized organic waste recovery and on-field application, although it can become difficult to utilize CHP waste heat and to keep transport requirements low.

Table 4: Carbon footprint metrics related to the treatment of 1 kg of food waste in the different scenarios

City	AD to insects ratio for primary waste	Carbon balance of "integrated" waste management	Carbon balance of incineration	Benefit of "integrated" waste management versus incineration
Glasgow	0.64	- 0.0225 kg CO_2e/kg FW	0.4141 kg CO_2e/kg FW	- 0.4366 kg CO_2e/kg FW
Lyon	0.90	0.0063 kg CO_2e/kg FW	0.4904 kg CO_2e/kg FW	- 0.4841 kg CO_2e/kg FW

4 Discussion

Implications of the results are discussed and a guide to transpose results to other cities, i.e. how spatial, population and climate factors influence the outcome, is provided. Relevant assumptions, sensitivities and limitations of this study are discussed and key learnings are then summarized. Comparisons with the results from other studies for some specific findings are discussed in H.3 and H.4 in the SI. It is important to note that the learnings from this study relate only to resource consumption and carbon footprints of urban farming. Other equally crucial considerations (e.g. economic opportunity, food and nutrition equity, higher affinity for plant-based diets, ecosystem services and disservices) of such systems have not been considered.

4.1 Implications of growth intensity

The variation of the ratio between greenhouse growing and urban gardening shows an inverse proportional relationship between self-sufficiency and system emissions with a clear trade-off. Further, the monthly self-sufficiency profile for both extremes lends insight into how a UA system may be shaped depending on how its socio-ecological benefits are weighed against the perspective of productivity and food security. Giving more emphasis to the former, options such as urban gardens would be preferred, leading to more inclusive development but also likely the need for storage and processing facilities. In contrast, if stabilized year-round production is more emphasized, the system would favor options such as high-intensity greenhouses, which come with a potentially much higher carbon footprint but a steady supply and thus simpler logistics. Another consideration is public perception, as more exclusive and less aesthetic greenhouses are likely undesired in many urban settings. Besides, the choice between these two types of options may be affected by the local circumstances. In general, a comprehensive assessment of the advantages and disadvantages of the various growing options would help decision makers to assess the necessary trade-off in the scale-up of urban agriculture.

4.2 Remarks on resource consumption

In the water use scenarios analyzed in this work, rainwater harvesting brings significant benefits up to the point of reducing the water input to zero when shared distribution and storage is assumed in a location with extensive rainfall. The difference in water demand between the cities is primarily caused by the difference in annual rainfall. Making use of low-tech hoop-houses with rainwater harvesting equipment can reduce water requirements for gardens by half, and can further minimize the external water requirements with some decentralized storage capacity. It could be claimed that if urban agriculture is to be scaled up, new infrastructure should incorporate rainwater harvesting facilities, although the possibility of sharing reservoirs and installing inter-monthly storage of rainwater would be somewhat limited. Technical aspects such as disinfection after longer storage or distribution losses are yet to be considered in future work.

Fertilizer use was greatly reduced through utilization of waste products. The use of chemical nitrogen fertilizer in soil-based growing in the “integrated” scenario occurred mainly due to the differences in plant availability assumed for the nutrients in compost and digestate fiber: The nitrogen had a very low plant availability, thus P and K became the constraining factors for further waste nutrient addition. In reality, different countries use varying fertilization rates of nitrogen for soil-based growing, and growers obtain similar crop yields whether grown with chemical fertilizer or with compost only (Cleveland, 1997), hence it may be projected that the waste products from decentralized waste conversion could make chemical fertilizer obsolete for soil-based growing. Hydroponic growing solutions on the other hand require exact dosing and fertilizer levels for maximum productivity, hence it is likely that growth is adversely affected by nutrient deficiency (e.g. reduced yields or change in taste). Therefore, considering more advanced nutrient recovery technologies capable of concentrated extraction of a single plant nutrient would allow to better tailor the nutrient supply to the plant needs in hydroponics.

This study likely overestimated the energy inputs and thus emissions for the yield figures employed: (i) a modelling study by Benis et al. (2017b) for Paris used higher PAR values (20 vs. 13.5 mol/m^2 in this work) and thus higher yields (75 versus 42.9 kg/m^2 in this work) while only requiring 93 kWh/m^2 of lighting (versus 143 kWh/m^2 in this work), (ii) a study for a vertical farm in Lyon only had a life cycle impact of 0.39 kg CO_2e/kg lettuce (Romeo et al., 2018) and (iii) an empirical study by Theurl et al. (2014) operated a greenhouse in Austria without artificial lighting while achieving similar yields to this work. The latter study also found that heating-related annual carbon footprint could be almost halved by not running the greenhouse for 1-2 months in winter (Theurl et al., 2014).

The more food is produced in urban environments, the higher the potential waste assimilation. Considering food waste alone, a large-scale UA system in a less dense city such as Glasgow could assimilate half the generated amount, which would equal either the share of household or commercial food waste, thus allowing for tailored collection strategies. In terms of closing the nutrient loop, it seems that a great amount of phosphorus would still be lost, calling for holistic solutions for recovery in a range of waste streams. From an urban metabolism perspective, a well-designed UA system in terms of the management of resources including water, nutrients and energy, as discussed above, has the potential to reduce material imports and their impacts for feeding the urban population while keeping the operational outputs to the environment (e.g. emissions to air) at least no worse than the current system.

4.3 Influence profile of city characteristics on outcomes

The differences between the two cities (see Table 2) have influenced the results significantly. Figure 9 shows how specific characteristics have influenced a certain outcome and enables transposing the results to other cities.

	Spatial			Climate			Other			
Outcome	Population density	green-space as % of urban area	Flat roof-space as % of urban area	Temperature	Solar radiation	Rainfall	CF of electricity grid[1]	Productivity level[2]	Ratio Digeponics / Aquaponics[3]	Ratio greenhouse / soil-based growing[4]
Heating intensity				- -	-					
Lighting intensity					↓					
Differences in seasonal production				(-)	(-)					+ +
Self-sufficiency F&V at 20% GH	↓	+ +	+							+ +
Self-sufficiency fish at 20% GH	↓	++							- -	+ +
Water use efficiency (SB, no harvesting)				-		+ +				
Fertilizer use reduction										+
Carbon intensity of system ("UA+Ext" and "integrated")				-	-		+ +			↑
Waste assimilation								↑	-	+ +
CHP-utilisation				-						+
P-recovery								↑		+
CF benefit assuming full waste product assimilation							-		-	

Figure 9: Influence matrix, meaning of symbols: ↑↓(approximately) proportional or inversely proportional, ++ -- strong influence (increasing or decreasing), +- weak influence (increasing or decreasing), () Expected, not calculated in this study

1 0.41 kg CO_2 eq/kWh for Glasgow and 0.06 kg CO_2 eq/kWh for Lyon
2 "Productivity level" is the chosen degree of self-sufficiency, hence every factor that influences self-sufficiency outcomes also affects the other outcomes marked in the table as being affected by "productivity level"
3 1.136 for Glasgow and 1.917 for Lyon
4 Ratio by weight for annual produce, 1.832 for Glasgow and 2.605 for Lyon

The integration benefit (percentage improvement, not shown) concerning heat and electricity consumption depends mainly on the ratio between absolute heating requirements (determined by the self-sufficiency and a high share of greenhouses) and the potential amount of heat generated. If the ratio of digeponic to aquaponics increases, a higher demand for digestate leads to a higher AD throughput, resulting in more biogas and thus energy generated but a lower larvae demand and thus reduced input replacement benefits. For the integration benefit associated with waste treatment, a lower ratio of greenhouses versus soil-based growing reduces secondary treatment requirements of the surplus digestate fiber. Further comparative insights related to the results can be found in the SI (H.5). As a general guide, the approach of this work is valid in cities where the architecture of buildings features low to medium heights with flat roofs, potential access and sufficient structural integrity, where urban greenspaces have clean or remediable soils and where some urban space is available for (semi-) decentral waste conversion. Prohibitive would be very dense urban areas, areas with high cultural sensitivities over waste utilization for food production and locations in the polar and subpolar climate zones as well as in arid or desert-like conditions. For sparsely populated cities or urban areas which include agricultural land, the possibility of enhanced land spreading may increase the sink potential for converted waste products. For such cases, the model may be adapted to include mechanized distribution and application of liquid or solid fertilizers.

4.4 Assumptions, uncertainties and limitations

This modelling exercise of relevant material and energy flows relied naturally on a range of assumptions and simplifications to allow for a quantitative assessment of the feasibility, desirability and sustainability of such integrated systems. For example, it was assumed that the carbon footprint of the logistics network for both, the distribution of food and management of waste, is the same in all scenarios and therefore not included in the assessment. While some argue that switching to source-separation of food waste increases emissions overall (Chiew et al., 2015), alternative logistic arrangements are available, such as aggregation at neighborhood collection points for semi-decentralized conversion facilities (Guo et al., 2018) or cargo-bike schemes for the collection of wastes (Walker et al., 2017), both with considerably lower footprint than household pick-up with large diesel trucks. A more direct delivery of urban produce, e.g. through bike couriers (Wrighton & Reiter, 2016), electric vans (Kulak et al., 2013), mobile markets (Widener et al., 2012) and local pick-up points (Opitz et al., 2017) could also reduce refrigeration and storage emissions. Further assumptions or simplifications have been applied to allow the modelling of this work, such as excluding among others the shading effects of buildings, the influence of the urban heat island effect, CO_2 enrichment, plant-specific soil-based fertilization rates, humidification, micronutrient availability, labor requirements, nutrient and waste run-off, and the germination and nursing processes.

In addition to the above assumptions, the values of a number of model parameters were fixed in this study, which in reality either (a) naturally fluctuate in time or (b) depend on the choice of material or system design. The choice of fixed values for these parameters inevitably introduces uncertainties to the results. For (a), the composition and amount of food and garden waste, which were fixed to representative values in the study, have a direct effect on the monthly nutrient balance and biogas yield. Their fluctuation would mainly influence the daily and to some extent the monthly material flow but would not distort the overall annual result displayed in figure 6. The climate dependent yield values have an effect on the expected productivity but the chosen range of crops and thus diversified yield figures likely capture reality better than estimates focusing only on one specific crop. A constant demand profile for fruit and vegetable and a fixed distribution of crops between growing practices were assumed for both cities, but the actual demand profiles can vary quite substantially throughout the year, and in reality the crop choice of the growers is most likely also to vary, influenced by economics and consumers' preference rather than assuring proportional demand satisfaction. These details remain to be captured in future studies.

For (b), a parameter with significant influence was the limit of area use set to the maximum scale-up. Naturally, if all applicable areas were used in this study the self-sufficiency and all related impacts would be proportionally higher. It could be claimed that the limits used in this work are already too high and unrealistic; they nevertheless serve as an indication and benchmark to allow comparison and facilitate discussions about the potential. On the energy balances of greenhouses, a strong effect on the results comes from the choice of construction material which determines the K-value (chosen as 2.27 $W/m^2/K$ in the model) for heat lost to the surroundings in greenhouses, with almost full proportionality for the resulting heating requirements. Other studies have used higher figures around 5-6 $W/m^2/K$ (Nadal et al., 2017) while novel types of greenhouses, e.g. with bubble-insulation, promise K-values as low as 0.9 $W/m^2/K$ (Stoknes et al. 2016). The yield values for greenhouse growing constitute another significant influence. Although these were selected based on reports from case studies, they may differ and thus almost proportionally increase or reduce the associated environmental burden per kg of produce. For example, a modelling study on rooftop greenhouses in Portugal with similar growing conditions achieved more than 1.7 times the yield for tomatoes assumed in this work (42.9 versus 74 kg/m^2)(Benis et al., 2017c).

On limitations in its scope, this work only considers the operational footprint, similar to many other studies (Barbosa et al., 2015; Benis et al., 2017a; Sanyé-Mengual et al., 2018; Shiina et al., 2011). Clearly structural components for greenhouses, building retrofits, waste conversion facilities and local gardens will have an environmental footprint. At the same time, the infrastructure for the existing global supply system is already in place and constantly refurbished, making the inclusion a complex exercise. In a study by Boulard et al. (2011) emissions from the raw materials and construction of a greenhouse for a 20-year project duration were included and found that they only contributed a small amount (0.14 kg CO_2e/kg product, or 6.8% of the total footprint) in comparison to the operational inputs such as heat and fertilizer. The validity of the outcomes of this work is therefore unlikely to be significantly affected by only considering the operational footprint. Overall, the learning from this study can be seen as indicative and providing first ballpark figures rather than aspiring to obtain fully conclusive results. Lastly, only the directly relevant share of the biomass flows (i.e. urban fruit and vegetable produce, fish, food and garden waste) within the cities were considered. These flows have only a marginal effect on the entire urban biomass metabolism. This is because the total biomass flows in-, out and within are much greater (Bahers & Giacchè, 2019) and include also processed foods and exports as well as other organic waste streams. Incorporating those and expanding the geographical scope to include the surrounding region may offer further opportunities for circular and holistic organic waste management (Dubbeling et al., 2016).

4.5 Key insights

The primary purpose of simulating and optimizing the organic waste valorization pathways for use in urban agriculture was to understand the associated material flows in relation to urban space and population characteristics and to find out which pathways are the most promising for reducing emissions. A few insights can be gathered from the outcomes:

1. The carbon footprints of the two UA scenarios are highly influenced by the adoption level of energy-intensive greenhouses growing in winter months which may result in a significant increase in emissions. At the same time, less intense lighting and heating schedules could reduce the operational footprint of greenhouses substantially. Further, fully renewable – and thus low-carbon – electricity grids in the future favor the UA scenarios in terms of GHG emissions. The advantages of a warmer and wetter climate with more sunshine can be seen in the resource intensity of high-intensity greenhouse production and make integrated UA and decentralized waste management systems more attractive in such climate conditions.

2. Even a fully-scaled up productive urban food system is unlikely to assimilate all the food and green waste generated in a city, as a large share of digestate products would not find an urban sink, even in less dense cities such as Glasgow. External or central treatment of organic waste is therefore always required, and matching the throughput of small-scale neighborhood AD facilities with local fertilizer requirements or the sizes of other local sinks could be the most sensible approach.

3. Insect rearing on food waste for fish feed was the most preferred pathway in the optimization. This renders a promising combination of technologies for increased protein self-sufficiency and reduced waste burden while likely adding economic opportunities. The direct feeding of larvae to fish, rather than the energy intensive drying and processing to powder (Salomone et al., 2017), makes live BSF larvae a low-footprint feed while the local production and fresh consumption of fish could reduce emissions during transport and storage (Ziegler et al., 2003). The required urban area is also minimal since the operations can be carried out in multi-story buildings.

4. Co-located small- and micro-scale anaerobic digestion is a strong contender for the decentralized treatment of food wastes if local sink capacity is available, as the produced biogas and biofertilizer has major environmental advantages over waste-to-energy schemes and composting. The localized utilization of heat during biogas combustion complemented by the injection of biomethane into a low pressure urban grid or fuel stations represents a major advantage of decentralized or co-located AD against centralized AD. Pick-up schemes and collaborations with businesses as currently investigated by an EU-funded project with pilot cases in Lyon could facilitate the waste logistics and create a stable AD feed supply (http://www.decisive2020.eu/). Given the space constraints in urban environments and community gardens, total digester volume would have to be distributed to a number of smaller units (e.g. 2-50 m^3) co-located with greenhouses, calling for more scattered locations of smaller AD facilities. Besides in greenhouses, AD facilities could also be co-located in gardens with adjacent heat sinks

(e.g. bakeries, community cafés) to ensure the benefits of CHP heat use (see Walker et al. (2017)).

5. The nutrient, feed and water requirements for urban growing operations could be met by a significant degree via integration with waste management and incorporating rainwater harvesting. If mainly open-air gardens, low-tech hoop-houses and a few greenhouses with less-intensive growing schedules (e.g. reducing supplemental lighting) are employed, well-designed urban agriculture and waste management systems mean no additional burden to the resource use and the carbon footprint profile of a city while contributing to a more resilient urban food system.

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References

Alshrouf, A. (2017). Hydroponics , Aeroponic and Aquaponic as Compared with Conventional Farming. *American Scientific Research Journal for Engineering, Technology, and Sciences*, *27*(1), 247–255.

Audsley, E., Brander, M., Chatterton, J., Murphy-Bokern, Webster, & Williams. (2009). *How Low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope reduction by 2050*.

Bahers, J. B., & Giacchè, G. (2019). Towards a metabolic rift analysis: The case of urban agriculture and organic waste management in Rennes (France). *Geoforum*, *98*(February 2018), 97–107. https://doi.org/10.1016/j.geoforum.2018.10.017

Baltzell, A. H. (2011). *The Suitability of Worm Castings & Compost Tea in Organic Hydroponic Lettuce Propagation*. University of Arizona.

Banks, C. J., Chesshire, M., Heaven, S., & Arnold, R. (2011). Anaerobic digestion of source-segregated domestic food waste: Performance assessment by mass and energy balance. *Bioresource Technology*, *102*(2), 612–620. https://doi.org/10.1016/j.biortech.2010.08.005

Barbosa, G. L., Almeida Gadelha, F. D., Kublik, N., Proctor, A., Reichelm, L., Weissinger, E., … Halden, R. U. (2015). Comparison of land, water, and energy requirements of lettuce grown using hydroponic vs. Conventional agricultural methods. *International Journal of Environmental Research and Public Health*, *12*(6), 6879–6891. https://doi.org/10.3390/ijerph120606879

Barles, S. (2010). Society, energy and materials: The contribution of urban metabolism studies to sustainable urban development issues. *Journal of Environmental Planning and Management*, *53*(4), 439–455. https://doi.org/10.1080/09640561003703772

Benis, K., Reinhart, C., & Ferrão, P. (2017a). Building-Integrated Agriculture (BIA) In Urban Contexts : Testing A Simulation-Based Decision Support Workflow. In *15th IBPSA Conference San*.

Benis, K., Reinhart, C., & Ferrão, P. (2017b). Building-Integrated Agriculture (BIA) In Urban Contexts: Testing A Simulation-Based Decision Support Workflow. *Journal of Cleaner Production*, *147*(August), 589–602. https://doi.org/10.1016/j.jclepro.2017.01.130

Benis, K., Reinhart, C., & Ferrão, P. (2017c). Development of a simulation-based decision support workflow for the implementation of Building-Integrated Agriculture (BIA) in urban contexts. *Journal of Cleaner Production*, *147*, 589–602. https://doi.org/10.1016/j.jclepro.2017.01.130

Boulard, T., Raeppel, C., Brun, R., Lecompte, F., Hayer, F., Carmassi, G., & Gaillard, G. (2011). Environmental impact of greenhouse tomato production in France. *Agronomy for Sustainable Development*, *31*(4), 757–777. https://doi.org/10.1007/s13593-011-0031-3

Bowman, J. (2017). Tradeoffs in aquaponics vs hydroponics, by the numbers. Retrieved February 7, 2019, from https://medium.com/edenworks/tradeoffs-in-aquaponics-vs-hydroponics-by-the-numbers-6d12b0ecd0af

Bugbee, B. (2004). Nutrient Management in Recirculating Hydroponic Culture. *Acta Horticulturae*, (648), 99–112. https://doi.org/10.17660/ActaHortic.2004.648.12

Chiew, Y. L., Spångberg, J., Baky, A., Hansson, P. A., & Jönsson, H. (2015). Environmental impact of recycling digested food waste as a fertilizer in agriculture - A case study. *Resources, Conservation and Recycling*, *95*(2015), 1–14. https://doi.org/10.1016/j.resconrec.2014.11.015

Cleveland, D. (1997). Are urban gardens an efficient use of resources? Retrieved August 8, 2018, from https://cals.arizona.edu/OALS/ALN/aln42/cleveland.html

Clinton, N., Stuhlmacher, M., Miles, A., Uludere Aragon, N., Wagner, M., Georgescu, M., … Gong, P. (2018). A Global Geospatial Ecosystem Services Estimate of Urban Agriculture. *Earth's Future*. https://doi.org/10.1002/2017EF000536

Colasanti, K., & Hamm, M. W. (2010). Assessing the local food supply capacity of Detroit, Michigan. *Journal of Agriculture, Food Systems, and Community Development*, *1*(2), 41–58. https://doi.org/10.5304/jafscd.2010.012.002

David, A. (2013). *Technical document on municipal solid waste organics processing*. Ottawa.

Deelstra, T. (1987). Urban Agriculture and the Metabolism of Cities. *Food and Nutrition Bulletin*, *9*(2), 1–3. https://doi.org/10.1177/156482658700900210

Dimambro, M., Steiner, J., & Rayns, F. (2015). *Literature review : Digestate use in protected horticulture*.

Dubbeling, Marielle, Bucatariu, Camelia, Santini, Guido, Vogt, Carmen, E. (2016). *City Region Food Systems and Food Waste Management*.

Fisch-Informationszentrum. (2016). *Fischwirtschaft - Daten und Fakten*. Hamburg. https://doi.org/10.1007/s11298-013-0792-x

Goldstein, B., Hauschild, M., Fernandez, J., & Birkved, M. (2016). Testing the environmental performance of urban agriculture as a food supply in northern climates. *Journal of Cleaner Production*, *135*, 984–994. https://doi.org/10.1016/j.jclepro.2016.07.004

Goldstein, B., Hauschild, M., Fernández, J., & Birkved, M. (2016). Urban versus conventional agriculture, taxonomy of resource profiles: a review. *Agronomy for Sustainable Development*, *36*(1), 1–19. https://doi.org/10.1007/s13593-015-0348-4

Goldstein, B. P., Hauschild, M. Z., Fernández, J. E., & Birkved, M. (2017). Contributions of Local Farming to Urban Sustainability in the Northeast United States. *Environmental Science and Technology*, *51*(13), 7340–7349. https://doi.org/10.1021/acs.est.7b01011

Grard, B. J.-P., Bel, N., Marchal, N., Madre, F., Castell, J.-F., Cambier, P., … Aubry, C. (2015). Recycling urban waste as possible use for rooftop vegetable garden. *Future of Food: Journal on Food, Agriculture and Society*, *3*(1), 21–34.

Grewal, S. S., & Grewal, P. S. (2012). Can cities become self-reliant in food? *Cities*, *29*(1), 1–11. https://doi.org/10.1016/j.cities.2011.06.003

Guo, W., Zhou, Y., Zhu, N., Hu, H., Shen, W., Huang, X., … Li, Z. (2018). On site composting of food waste: A pilot scale case study in China. *Resources, Conservation and Recycling*, *132*(September 2017), 130–138. https://doi.org/10.1016/j.resconrec.2018.01.033

H Adwiraah, S Oldenburg, I Körner, R. S. (2012). Bioresources inventory. GIS-based identification, quantification and characterisation methods. *ORBIT*, 80–87.

Haberman, D., Gillies, L., Canter, A., Rinner, V., Pancrazi, L., & Martellozzo, F. (2014). The Potential of Urban Agriculture in Montréal: A Quantitative Assessment. *ISPRS International Journal of Geo-Information*, *3*(3), 1101–1117. https://doi.org/10.3390/ijgi3031101

Health Canada. (n.d.). Canada's Food Guide. Retrieved November 26, 2018, from https://www.canada.ca/content/dam/hc-sc/migration/hc-sc/fn-an/alt_formats/hpfb-dgpsa/pdf/food-guide-aliment/view_eatwell_vue_bienmang-eng.pdf

Joly, G. (2018). *Valorising Organic Waste using the Black Soldier Fly (Hermetia illucens), in Ghana*. KTH Royal Institue of Technology, Stockholm.

Kawamura-Aoyama, C., Fujiwara, K., Shinohara, M., & Takano, M. (2014). Study on the hydroponic culture of lettuce with microbially degraded solid food waste as a nitrate source. *Japan Agricultural Research Quarterly*, *48*(1), 71–76. https://doi.org/10.6090/jarq.48.71

Koegler, M., Jean-Paul, G. B., & Aubry, C. (2017). *Climate Innovation Potentials of Urban Agriculture (CIPUrA) Geographic Pathfinder*.

Kulak, M., Graves, A., & Chatterton, J. (2013). Reducing greenhouse gas emissions with urban agriculture: A Life Cycle Assessment perspective. *Landscape and Urban Planning*, *111*(1), 68–78. https://doi.org/10.1016/j.landurbplan.2012.11.007

López-Mosquera, M. E., Fernández-Lema, E., Villares, R., Corral, R., Alonso, B., & Blanco, C. (2011). Composting fish waste and seaweed to produce a fertilizer for use in organic agriculture. *Procedia Environmental Sciences*, *9*, 113–117. https://doi.org/10.1016/j.proenv.2011.11.018

Love, D. C., Fry, J. P., Li, X., Hill, E. S., Genello, L., Semmens, K., & Thompson, R. E. (2015). Commercial aquaponics production and profitability: Findings from an international survey. *Aquaculture*, *435*, 67–74. https://doi.org/10.1016/j.aquaculture.2014.09.023

Mohareb, E., Heller, M., Novak, P., Goldstein, B., Fonoll, X., & Raskin, L. (2017). Considerations for reducing food system energy demand while scaling up urban agriculture. *Environmental Research Letters*, *12*(12). https://doi.org/10.1088/1748-9326/aa889b

Mugica, Y., & Spacht, A. (2017). *San Francisco Composting from fork to farm and back*. San Francisco.

Nadal, A., Alamús, R., Pipia, L., Ruiz, A., Corbera, J., Cuerva, E., … Josa, A. (2017). Urban planning and agriculture. Methodology for assessing rooftop greenhouse potential of non-residential areas using airborne sensors. *Science of the Total Environment*, *601–602*, 493–507. https://doi.org/10.1016/j.scitotenv.2017.03.214

Opitz, I., Doernberg, A., Specht, K., & Piorr, A. (2017). Future|Food|Commons: Alternative Food Networks at the urban-rural interface, (July).

Pantanella, E., Cardarelli, M., Colla, G., Rea, E., & Marcucci, A. (2012). Aquaponics vs. hydroponiics: production and quality of lettuce crop. In *Acta Horticulturae* (pp. 887–893). International Society for Horticultural Science (ISHS), Leuven, Belgium. https://doi.org/10.17660/ActaHortic.2012.927.109

Romeo, D., Vea, E. B., & Thomsen, M. (2018). Environmental Impacts of Urban Hydroponics in Europe: A Case Study in Lyon. *Procedia CIRP*, *69*(May), 540–545. https://doi.org/10.1016/j.procir.2017.11.048

Saha, M., & Eckelman, M. J. (2017). Growing fresh fruits and vegetables in an urban landscape: A geospatial assessment of ground level and rooftop urban agriculture potential in Boston, USA. *Landscape and Urban Planning*, *165*(August 2016), 130–141. https://doi.org/10.1016/j.landurbplan.2017.04.015

Salomone, R., Saija, G., Mondello, G., Giannetto, A., Fasulo, S., & Savastano, D. (2017). Environmental impact of food waste bioconversion by insects: Application of Life Cycle Assessment to process using Hermetia illucens. *Journal of Cleaner Production*, *140*, 890–905. https://doi.org/10.1016/j.jclepro.2016.06.154

Sanjuan-Delmás, D., Llorach-Massana, P., Nadal, A., Ercilla-Montserrat, M., Muñoz, P., Montero, J. I., … Rieradevall, J. (2018). Environmental assessment of an integrated rooftop greenhouse for food production in cities. *Journal of Cleaner Production*, *177*, 326–337. https://doi.org/10.1016/j.jclepro.2017.12.147

Sanyé-Mengual, E., Martinez-Blanco, J., Finkbeiner, M., Cerdà, M., Camargo, M., Ometto, A. R., … Rieradevall, J. (2018). Urban horticulture in retail parks: Environmental assessment of the potential implementation of rooftop greenhouses in European and South American cities. *Journal of Cleaner Production*, *172*, 3081–3091. https://doi.org/10.1016/j.jclepro.2017.11.103

Shao, Y., Heath, T., & Thu, Y. (2016). Developing an economic estimation system for vertical farms. *International Journal of Agricultural and Environmental Information Systems*, *7*(2).

Shiina, T., Hosokawa, D., Roy, P., Nakamura, N., Thammawong, M., & Orikasa, T. (2011). Life cycle inventory analysis of leafy vegetables grown in two types of plant factories. In *Acta Horticulturae* (pp. 115–122). International Society for Horticultural Science (ISHS), Leuven, Belgium. https://doi.org/10.17660/ActaHortic.2011.919.14

Shinohara, M., Aoyama, C., Fujiwara, K., Watanabe, A., Ohmori, H., Uehara, Y., & Takano, M. (2011). Microbial mineralization of organic nitrogen into nitrate to allow the use of organic fertilizer in hydroponics. *Soil Science and Plant Nutrition*, *57*(2), 190–203. https://doi.org/10.1080/00380768.2011.554223

Sioen, G. B., Sekiyama, M., Terada, T., & Yokohari, M. (2017). Post-disaster food and nutrition from urban agriculture: A self-Sufficiency analysis of Nerima ward, Tokyo. *International Journal of Environmental Research and Public Health*, *14*(7), 1–20. https://doi.org/10.3390/ijerph14070748

Sioen, G. B., Terada, T., Sekiyama, M., & Yokohari, M. (2018). Resilience with mixed agricultural and urban land uses in Tokyo, Japan. *Sustainability (Switzerland)*, *10*(2), 1–27. https://doi.org/10.3390/su10020435

Stoknes, K., Scholwin, F., Krzesiński, W., Wojciechowska, E., & Jasińska, A. (2016). Efficiency of a novel “Food to waste to food” system including anaerobic digestion of food waste and cultivation of vegetables on digestate in a bubble-insulated greenhouse. *Waste Management*, *56*, 466–476. https://doi.org/10.1016/j.wasman.2016.06.027

Theurl, M. C., Haberl, H., Erb, K. H., & Lindenthal, T. (2014). Contrasted greenhouse gas emissions from local versus long-range tomato production. *Agronomy for Sustainable Development*, *34*(3), 593–602. https://doi.org/10.1007/s13593-013-0171-8

Walker, M., Theaker, H., Yaman, R., Poggio, D., Nimmo, W., Bywater, A., … Pourkashanian, M. (2017). Assessment of micro-scale anaerobic digestion for management of urban organic waste: A case study in London, UK. *Waste Management (New York, N.Y.)*, *61*, 258–268. https://doi.org/10.1016/j.wasman.2017.01.036

Weidner, T., Yang, A., & Hamm, M. W. (2019). Consolidating the current knowledge on urban agriculture in productive urban food systems: Learnings, gaps and outlook. *Journal of Cleaner Production*, *209*, 1637–1655. https://doi.org/10.1016/J.JCLEPRO.2018.11.004

White, J. T., & Bunn, C. (2017). Growing in Glasgow: Innovative practices and emerging policy pathways for urban agriculture. *Land Use Policy*, *68*(July), 334–344. https://doi.org/10.1016/j.landusepol.2017.07.056

Widener, M. J., Metcalf, S. S., & Bar-Yam, Y. (2012). Developing a mobile produce distribution system for low-income urban residents in food deserts. *Journal of Urban Health*, *89*(5), 733–745. https://doi.org/10.1007/s11524-012-9677-7

Wielemaker, R. C., Weijma, J., & Zeeman, G. (2018). Harvest to harvest: Recovering nutrients with New Sanitation systems for reuse in Urban Agriculture. *Resources, Conservation and Recycling*, *128*, 426–437. https://doi.org/10.1016/j.resconrec.2016.09.015

Wrighton, S., & Reiter, K. (2016). CycleLogistics – Moving Europe Forward! *Transportation Research Procedia*, *12*, 950–958. https://doi.org/10.1016/J.TRPRO.2016.02.046

Zhang, S., Bi, X. T., & Clift, R. (2013). A Life Cycle Assessment of integrated dairy farm-greenhouse systems in British Columbia. *Bioresource Technology*, *150*, 496–505. https://doi.org/10.1016/j.biortech.2013.09.076

Ziegler, F., Nilsson, P., Mattsson, B., & Walther, Y. (2003). Life Cycle assessment of frozen cod fillets including fishery-specific environmental impacts. *The International Journal of Life Cycle Assessment*, *8*(1), 39–47. https://doi.org/10.1007/BF02978747