Evaluating the sustainability and resiliency of local food systems

Nicole Josiane Janet Kennard

A thesis submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy

The University of Sheffield
Faculty of Science
Department of Chemistry

January, 2023

for the love of food

Abstract

With an ever-rising global population and looming environmental challenges such as climate change and soil degradation, it is imperative to increase the sustainability of food production. The drastic rise in food insecurity during the COVID-19 pandemic has further shown a pressing need to increase the resiliency of food systems. One strategy to reduce the dependence on complex, vulnerable global supply chains is to strengthen local food systems, such as by producing more food in cities. This thesis uses an interdisciplinary, food systems approach to explore aspects of sustainability and resiliency within local food systems.

Lifecycle assessment (LCA) was used to evaluate how farm scale, distance to consumer, and management practices influence environmental impacts for different local agriculture models in two case study locations: Georgia, USA and England, UK. Farms were grouped based on urbanisation level and management practices, including: urban organic, peri-urban organic, rural organic, and rural conventional. A total of 25 farms and 40 crop lifecycles were evaluated, focusing on two crops (kale and tomatoes) and including impacts from seedling production through final distribution to the point of sale. Results were extremely sensitive to the allocation of composting burdens (decomposition emissions), with impact variation between organic farms driven mainly by levels of compost use. When composting burdens were attributed to compost inputs, the rural conventional category in the U.S. and the rural organic category in the UK had the lowest average impacts per kg sellable crop produced, including the lowest global warming potential (GWP). However, when subtracting avoided burdens from the municipal waste stream from compost inputs, trends reversed entirely, with urban or peri-urban farm categories having the lowest impacts (often negative) for GWP and marine eutrophication. Overall, farm management practices were the most important factor driving environmental impacts from local food supply chains.

A soil health assessment was then performed on a subset of the UK farms to provide insight to ecosystem services that are not captured within LCA frameworks. Better soil health was observed in organically-farmed and uncultivated soils compared to conventionally-farmed soils, suggesting higher ecosystem service provisioning as related to improved soil structure, flood mitigation, erosion control, and carbon storage. However, relatively high heavy metal concentrations were seen on urban and peri-urban farms, as well as those located in areas with previous mining activity. This implies that there are important services and disservices on farms that are not captured by LCAs.

Zooming out from a focus on food production, a qualitative methodology was used to explore experiences of food insecurity and related health and social challenges during the COVID-19 pandemic. Fourteen individuals receiving emergency food parcels from a community food project in Sheffield, UK were interviewed. Results showed that maintaining food security in times of crisis requires a diverse set of individual, household, social, and place-based resources, which were largely diminished or strained during the pandemic. Drawing upon social capital and community support was essential to cope with a multiplicity of hardship, highlighting a need to develop community food infrastructure that supports ideals of mutual aid and builds connections throughout the food supply chain. Overall, this thesis shows that a range of context-specific solutions are required to build sustainable and resilient food systems. This can be supported by increasing local control of food systems and designing strategies to meet specific community needs, whilst still acknowledging a shared global responsibility to protect ecosystem, human, and planetary health.

Acknowledgements

This thesis would not have been possible without the invaluable support, guidance, and encouragement of many others. Firstly, I would like to thank my PhD supervisors at the University of Sheffield – Tony Ryan, Duncan Cameron, and Jill Edmondson – for their openness and support to bring my research ideas to life, and their crucial expertise and guidance to keep me grounded throughout the process. I am particularly thankful for their understanding and encouragement to broaden my experiences outside my PhD, which has been instrumental in my overall development as a researcher and individual. I am also thankful to Megan Blake, who provided invaluable guidance, support, and supervision for the qualitative research within this thesis. I am additionally grateful to my colleagues and the many staff members within the Departments of Chemistry and Biosciences, who have provided extensive assistance and support throughout these four years.

I would also like to thank the Grantham Centre for Sustainable Futures at the University of Sheffield for providing me with financial support to complete this PhD, as well as numerous opportunities and trainings that have expanded my perspective and truly shaped me as a researcher. On the same note, I would like to extend my thanks to the Postgraduate Research Experience Programme at Sheffield, which funded me to embark upon several opportunities to explore interests outside my PhD.

This research would also not have been possible without the support of the individuals and organisations that graciously gave their time to these projects. Foremostly, I would like to extend my heartfelt thanks to the farmers who participated and generously donated their time and wisdom. Equally, I would like to thank the interview participants in Sheffield who took the time to share their experiences during the COVID-19 pandemic. The insight of the participants in these research projects has been invaluable in creating a work that reflects true context and experience. I am also thankful to the Foodhall Project and its volunteers for their support of this research; their tireless efforts to distribute food during the pandemic; and for the many meals, conversations, and friendships made and enjoyed within the space. I am also grateful to the Georgia Fruit & Vegetable Association; the University of Georgia Cooperative Extension; the Atlanta Farmers Coalition; the British Tomato Growers' Association; and the Brassica Growers' Association, who helped advertise the research studies.

Finally, I would like to thank my family and friends, both near and far, who have supported me throughout the many high's and low's of this journey. Their encouragement and understanding have been an unwavering source of strength, and I could not have done it without them. I extend my gratitude to my 'UK family', who have provided me with a home away from home: to Robert Bamford who gave invaluable insight, spent countless hours on country roads and in fields, and provided unwavering support that has truly been a lifeblood for this PhD; to Annabel Smith, whose overwhelming generosity and positive encouragement has always kept me going; and to Andrew White, who brought a piece of home to the UK and provided me with comfort and support throughout this journey.

Also, to my friends and family back home, who have been a source of sanity even an ocean away. Specifically, to my father, for first instilling my love of nature on all those Smokey Mountain mornings. To my Mimi, for her unconditional care, love, and positivity. And to my mom, whose strength, determination, and selflessness never ceases to amaze me. Thank you.

Declaration and list of publications and research outputs

I hereby declare that except where specific reference is made to the work of others, the contents of this thesis are original and have not been submitted in whole or in part for consideration for any other degree or qualification in this or any other university. I, the author, am aware of the University's Guidance on the Use of Unfair Means (www.sheffield.ac.uk/ssid/unfair-means) and confirm that this Thesis is my own work.

This research involved human participants and was given ethical clearance by the University of Sheffield ethics committee (application numbers 030300 and 035915).

It should be noted that the qualitative research as presented in Chapter 5 was undertaken with the support and supervision of Dr Megan Blake. Together, the results of this research were analysed, evaluated, and discussed, with an initial co-authored publication drafted, but not yet submitted. Thus, it should be noted that Chapter 5 includes outputs from this publication draft, but the main writing (as well as the data collection and project conceptualisation) was performed by myself as the author of this thesis.

Publications relevant to thesis

Other portions of the research in this thesis have been published in academic journals or conference proceedings during the time of this PhD. At the time of submission, this includes the following publications, which are referenced within this thesis where applicable. Conceptualisation and writing of each publication was primarily undertaken by the first author, where multiple authors are listed.

- Kennard, N.J. and Bamford, R.H. (2020) 'Urban Agriculture: Opportunities and Challenges for Sustainable Development', in Leal Filho, W. et al. (eds) *Zero Hunger. Encyclopedia of the UN Sustainable Development Goals*. Cham: Springer, pp. 929–942. doi:10.1007/978-3-319-95675-6_102.

- Kennard, N.J. (2020) 'Comparing resource use for tomato production on urban, peri-urban and rural farms in Georgia, USA', in *Urban Food Systems Symposium 2020*. Kansas State University Libraries: New Prairie Press.

Other research outputs relevant to thesis

Other written research outputs that were not published in academic journals were also produced during this PhD. This includes two contributions of written evidence to the UK Parliamentary inquiry on COVID-19 and food supply as conducted by the Environment, Food and Rural Affairs Committee, as well as a Parliamentary briefing about impacts of pesticides on health, which was written during a three month fellowship undertaken with the Parliamentary Office of Science and Technology in 2021 (see: Appendix C for more detail). A public-facing summary report of the study on barriers to accessing food during COVID-19 (Chapter 5) was also produced with the intention to share with community food groups; this report was also shared with the Sheffield City Council to inform their 2021 inquiry on food poverty. These research outputs are referenced in the following list. Where multiple authors are listed, writing was primarily undertaken by the first author.

- Kennard, N.J. and Tendler, I. (2020) *Written evidence submitted by The Foodhall Project*, Parliamentary inquiry on COVID-19 and food supply, Environment, Food and Rural Affairs Committee. London, UK. Available at: https://committees.parliament.uk/writtenevidence/3319/html/

- Kennard, N.J. (2021) *Follow-up written evidence,* Parliamentary inquiry on COVID-19 and food supply, Environment, Food and Rural Affairs Committee. London, UK. Available at: https://committees.parliament.uk/writtenevidence/21848/html/

- Kennard, N.J. (2023) 'Experiences accessing food during COVID-19: A case study with Foodhall, Sheffield.' The University of Sheffield. Public summary report. Available at: https://doi.org/10.15131/shef.data.23694957.v2

- Kennard, N.J. and Vagnoni, C. (2021) *Pesticides and health: POSTbrief 43*. London, UK. Available at: https://post.parliament.uk/research-briefings/post-pb-0043/.

Intended future publications from thesis (in drafting stages)

There are also several publications relevant to this thesis research that are currently in various stages of drafting. These articles are intended to be submitted to academic journals and published in the future. The following list provides an overview of these intended future publications. The preliminary list of authors for each is not finalised or exhaustive.

- Kennard, N.J., Ryan, A.J., and Cameron, D.D. (n.d.) 'Reconsidering sustainable agriculture: the importance of evaluating individual farm impacts.'
  This will relay the main results of the lifecycle assessment study presented in Chapter 3, comparing environmental impacts between different local agriculture models and describing the main factors that drive these differences.

- Kennard, N.J., Ryan, A.J., and Cameron, D. (n.d.) 'How should compost production burdens be allocated to farms?'
  This will draw upon lifecycle assessment results for different allocation scenarios as presented in Chapter 3. It will contribute to the development of lifecycle assessment methodology for food and farming applications by highlighting how different allocation decisions can significantly influence the impact results for farms.

- Kennard, N.J., Ryan, A.J., and Cameron, D. (n.d.) 'Critique of LCA for evaluating environmental sustainability on farms.'
  This publication will draw upon results from Chapters 3 and 4 to critique the use of LCA as a method to compare environmental sustainability between different farming systems. It will show the importance of testing for other ecosystem services to provide a more holistic view of sustainability.

- Kennard, N.J. and Blake, M. (n.d.) 'The multidimensional experience of food insecurity during crisis: insights from COVID-19.'
  This publication will draw upon results in Chapter 5 to provide an overview of the lived experience of food insecurity during COVID-19 in Sheffield. The experiences of participants

are used to highlight the types of individual, familial, social, and place-based resources needed to be food secure during times of crisis.

- Kennard, N.J. and Blake, M. (n.d.) ‘The role of community responses in addressing food insecurity during COVID-19.’
  This will draw upon results in Chapter 5 to critique the types of food and financial support offered to people living on low incomes during COVID-19 in the UK, focusing on cases where informal community responses were necessary to fill gaps in government efforts.

Table of contents

List of tables

List of figures

List of equations

Chapter 1: Introduction

This thesis investigates aspects of sustainability and resiliency within local food systems, drawing upon case studies in the United States (U.S.) and United Kingdom (UK). This is facilitated by the use of an interdisciplinary framework that allows for various environmental and social elements of the food system to be examined. In particular, this thesis is comprised of three main studies: a comparison of environmental impacts across different types of local farms in the U.S. and UK; a soil health and ecosystem service assessment on a subset of the UK farms; and a qualitative study that identifies barriers to food security and types of community support that can enable resiliency during a time of crisis.

Within this introductory chapter, a basic overview of the concepts and challenges surrounding food system sustainability and resiliency are first provided in Section 1.1. Further introductory material is then presented for the three main studies of this thesis (Sections 1.2-1.4, respectively), highlighting current literature and identifying research gaps that are addressed within this body of work. This introduction was written by first drawing upon a literature scoping assessment that informed the development of this PhD research, focused on local food systems and urban agriculture; this initial assessment was first provided in a Confirmation Review report submitted to the University of Sheffield in July 2019 and was also used to inform the writing of a chapter on urban agriculture for the Encyclopedia of the UN Sustainable Development Goals (Kennard and Bamford, 2020). Further introductory material for specific research projects as presented in this thesis was gathered and reviewed during the time these projects took place, as well as during the final writing stages of the thesis (see: Section 1.6 for more information on the research timeline).

1.1 Food systems

Throughout this research, the 'food system' is seen as a complex network that includes all the people, activities, resources, and products involved in the production, processing, distribution, consumption, and disposal of food (Neff, Chan and Smith, 2009). For example, this includes activities that happen throughout the food supply chain, such as growing and harvesting crops or raising animals, as well as activities involving food processing, packaging, transport, sale, and preparation. However, the systems view of food expands this approach by also including the policies that affect the food supply chain, the businesses and people involved, and other social, economic, and environmental factors that influence the way food is produced, distributed, and consumed (Horton *et al.*, 2017; Parsons, Hawkes and Wells, 2019). The scope of a food system may be defined based on local, regional, national, and global contexts.

The systems approach captures the complexity of the processes actually involved in ensuring food for a growing global population (Zurek *et al.*, 2018). It allows for relationships to be observed between different components of the food supply chain and the political, economic, social, and literal environment in which they sit (Neff, Chan and Smith, 2009). This allows for the consideration of how changes in one part of the food system may impact others and any trade-offs that may arise (Horton *et al.*, 2017). The food systems approach is interdisciplinary in nature (Parsons, Hawkes and Wells, 2019); indeed, the study of food, and the processes and issues related to it, touches almost all major disciplines – including the sciences, engineering, medicine, geography, economics, politics, cultural studies, and even literature and history. Thus, taking a food systems approach *requires* an interdisciplinarity approach, which is why this thesis focuses on an incorporation of engineering, scientific, and social science methods to assess the sustainability and resiliency of local food systems.

1.1.1 Aims and functions of the food system

At first glance, the main aim of the food system is obvious – to meet the nutritional needs of people so that they can survive. However, from a societal standpoint, what people and governments want from the food system is multifaceted – to provide a consistent supply of nutritious and culturally appropriate food, ensure that people can access, afford, and consume it, promote animal welfare and human health, build a competitive and socially-balanced agri-food sector, provide good jobs for people, minimise environmental impacts, protect and conserve the landscape, and contribute to climate change mitigation via carbon storage, among other cultural, social, and environmental services depending on the context (Hodbod and Eakin, 2015; Horton *et al.*, 2017; Zurek *et al.*, 2018; Hebinck *et al.*, 2021). Indeed, these are some of the main aims highlighted in the independent 'National Food Strategy' report commissioned by the UK Government (Dimbleby, 2021), as well as in the UK Government's own food strategy produced in response (Defra, 2022c). In these strategies, the agri-food sector is being called upon to not only reduce greenhouse gas emissions and environmental impacts to meet the Government's net-zero targets by 2050 (BEIS, 2021c), but also contribute positively to nature, provide better-paying and high-quality jobs, provide more affordable food to people, and reduce diet-related illnesses. It can be seen that as farmers are being called upon to produce more food for a growing population, they are also being asked to produce a range of other ecosystem services (Dimbleby, 2021).

Thus, the food system is clearly a complex and multifunctional entity, contributing to a variety of economic, environmental, and social aims depending on the specific local context. Many of the aims of the food system can be seen as contributing to two main concepts: food security and ecosystem services.

1.1.1.1 Food security

Food security is one of the main goals of any food system. Perhaps one of the most common definitions of food security is that provided during the 1996 World Food Summit, described as: "when all people, at all times, have physical and economic access to sufficient safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life" (FAO, 2008). The Food and Agriculture Organization (FAO) of the United Nations further identifies four interlinked dimensions that comprise food security, which must all be met consistently and simultaneously (FAO, 2008). These include: the physical availability of food locally; the ability to access that food both physically and financially; the ability to utilise that food, including receiving appropriate nutrients; and the consistency and stability of the previous three dimensions over time. In a global scope, food security is thus related to global food supply and availability, supply chain resilience, and issues of food safety during trade (Defra, 2021b). On a household scale, it is related to whether one can afford food, access nutritious and appropriate food (related to available shops and transportation), prepare and eat the food that is available, and obtain healthy and nutritious diets (Defra, 2021b). Food *in*security thus describes the state when these conditions are not met, often characterised by being unable to eat balanced meals, having to cut the size of meals, or skipping meals entirely (Coleman-Jensen, Alisha Rabbitt, Gregory and Singh, 2022).

1.1.1.2 Ecosystem services

In addition to providing a safe, nutritious, affordable, and accessible supply of food for people, the agri-food sector is being called upon to also provide a wide range of other ecosystem services (IPBES, 2018; Dimbleby, 2021). Indeed, UK agricultural policy is now incentivising farmers and land managers to provide various ecosystem services by using ‘public money to pay for public goods’ (Defra, 2020).

Many studies have attempted to define and categorise ‘ecosystem services,’ (de Groot, 1992; Costanza *et al.*, 1997; Daily, 1997; Noël and O’Connor, 1998), but perhaps one of the simplest and most used definitions is that provided by the Millennium Ecosystem Assessment (2003), which defines ecosystem services as “the benefits people obtain from ecosystems.” This assessment then classifies ecosystem services into four categories: provisioning services (products obtained from ecosystems, such as food, fuel, and fibre); regulating services (regulation of ecosystem processes, such as climate, water, and disease); cultural services (non-material benefits obtained from ecosystems, such as recreation, education, and aesthetic benefits); and supporting services, which are necessary for the production of all other ecosystem services, such as soil formation, habitat formation, and nutrient cycling (Millennium Ecosystem Assessment, 2003).

1.1.2 Sustainable food systems

‘Sustainability’ is a word commonly used across a variety of disciplines, but it has become muddled in use and can be quite vague, in some cases causing it to be rejected as having any real meaning (Monto, Ganesh and Varghese, 2005). In a basic definition, the word relates to the ability to maintain in the long-term. This can be seen as a specific practice, product, or policy that is ‘long-lasting’ (Atkinson *et al.*, 1997; Bonevac, 2010) or has the ‘capacity for continuance’ (Shaw and Newby, 1998). Bonevac (2010) sets the time frame for what is considered as ‘long-lasting’ as one in which people can reasonably claim to have justified beliefs about the specific topic or practice of interest, being 100-200 years at most.

The word ‘sustainability’ came to use in the environmental sphere with what is known as the ‘Brundtland Report’, published by the UN World Commission on Environment and Development (chaired by Gro Harlem Brundtland, a former Prime Minister of Norway) (Brundtland, 1987). This report included the first widely-recognised definition of sustainable development, which was described as: “development that meets the needs of the present without compromising the ability of future generations to meet their own needs.” This definition has also been applied to describe a sustainable food system as one in which healthy food is provided to meet current needs, whilst healthy ecosystems are maintained so that they can also support food production for future generations (Story, Hamm and Wallinga, 2009).

Although some have criticised the Brundtland definition of sustainable development because of the focus on ‘needs,’ which are not concrete and change over time (Goldin and Winters, 1995), this author contends that it is necessary to define the aims of (or what is needed from) a specific practice or policy in order to evaluate if it is ‘sustainable’, even if this changes over time; indeed, sustainable to what aim? Thus, what a ‘sustainable food system’ looks like depends on what a specific community, government, or group wants from the system (Zurek *et al.*, 2018) – only then can it be assessed to see if the system is able to meet these goals consistently over the long-term. Even when considering the most basic goal of the global food system – to ensure food security – providing this over the long-term requires other environmental, social, and economic elements, which is why these are now commonly considered to be a main function of the food system (see: Section 1.1.1). As future food security is threatened by crises such as climate change, resource depletion, soil degradation, increased disease, political conflict, and rising food prices, among many others, a sustainable food system must be able to minimise environmental impacts, protect and conserve soil, land, and water resources, remain economically viable whilst also supporting the livelihoods of those involved, and ensure sufficient nutrition and human health; without meeting these aims, there will not be the natural, financial, or human capital left to provide food for future generations.

At the same time, the ethics of sustainability within the context of the Brundtland definition has been called into question due to this very issue of defining the 'needs' or 'aims' of a specific population (Bonevac, 2010). For example, if sustainable development means meeting the needs of the present – then the needs of whom specifically? And at the cost to whom else?

In light of this issue, the concept of sustainability has now expanded from more environmental or economic framings to also include social justice and equity, maintaining that the food system should serve the needs of *all* and generally support the *common good* (Hebinck *et al.*, 2021). Social justice relates to a vision of society in which all members are physically and psychologically safe and secure (Adams, Bell and Griffin, 1997), or one which meets basic human needs, provides freedom from exploitation and oppression, and enables access to opportunity and participation (Allen, 2008). Thus, food security is seen a critical part of social justice, as food is a basic human need; conversely, for all to be food secure also requires social justice (Sumner, 2011). This view is furthered by the stipulation of access to adequate food as a basic human right (De Schutter, 2014), which has been recognised by the UN and maintained within legally-binding international law, specifically the 1966 International Covenant on Economic, Social and Cultural Rights (UN General Assembly, 1966); this has been signed by a total of 175 states (OHCHR, 2023), including the UK (ratified) and U.S. (signatory), although these countries still have not written this into their own constitutions or national law.

Thus, a sustainable food system is now seen as one which incorporates food security, ecosystem and human health, and justice elements (Story, Hamm and Wallinga, 2009; Sumner, 2011). For example, Hebinck et al., (2021) describes four universally-applicable societal goals for a sustainable food system, or indeed four requirements of the food system, as: "1) Healthy, adequate, and safe diets for all; 2) Clean and healthy planet; 3) Economically thriving food systems supportive of the common good; and 4) Just, ethical, and equitable food systems." These four themes are used as the main pillars of food system sustainability within this thesis. Consequently, throughout this research, a sustainable food system is seen as one that is equitable and just, where all have an ensured right to food that enables food security to be ubiquitously and consistently achieved; as such, a sustainable food system must protect and conserve natural resources and support the livelihoods of all involved so that this can be achieved in the long-term.

1.1.3 Resilient food systems

The concept of resiliency is derived from the fields of engineering and ecology, and focuses on how a system is able to respond to disturbance, i.e., shocks and stresses (Holling, 1996). In the engineering field, resilience focuses on the ability to resist disturbance and return to the prior steady-state of equilibrium (Holling, 1996); the ecological framing expands this to include the possibility of the system reaching a new and different state of equilibrium (Holling, 1973). Applications of the concept of resilience to food systems have focused on a social-ecological framing, similar to food system sustainability (Hodbod and Eakin, 2015; Doherty *et al.*, 2019). As such, the concept of food system resiliency relates to the ability of the system, in which humans and the environment are interlinked, to absorb and adapt to shocks and stresses and then return to a steady state, which may be the same or different from the original state (Llanos and Border, 2020). Although the terms ‘resiliency’ and ‘sustainability’ are often used interchangeably, it is more accurate to see resilience as a component of sustainability (Doherty *et al.*, 2019). Indeed, if a system is to be maintained in the long-term (i.e., be ‘sustainable’), then at some point this will likely require the system to be able to respond and adapt to shocks and stresses.

Zurek et al. (2022) further highlights three main components of food system resilience as the ‘three R’s’: robustness, recovery, and reorientation, all of which are supported by adaptation. Robustness refers to the system’s ability to resist disruption; recovery to the ability to return back to a steady-state after disruption; and reorientation to the ability to accept alternative states or outcomes, either before or after disruption. The latter is particularly related to food system transformation, for example, by changing the aims or demands of the food system to reduce vulnerability to shocks and stresses (e.g., ensuring ecosystem and human health); Zurek et al. (2022) highlights this as what is likely the most important pathway toward building food system resilience. Finally, it should be noted that the resiliency of the food system requires resiliency throughout, not just at certain parts of the supply chain; resiliency cannot be attributed to any one part of the system without understanding how the change in one part influences others (Doherty *et al.*, 2019).

The concept of food system resiliency has gained much attention in recent years since the outbreak of the COVID-19 pandemic in late 2019 (Garnett, Doherty and Heron, 2020; Bellamy *et al.*, 2021; Di Marcantonio, Twum and Russo, 2021; Zurek *et al.*, 2022). The pandemic keenly exposed the fragility and inflexibility of the global food system, as national food supply chains were at first unable to adequately respond to changes in demand that came with closures of schools, restaurants, and the hospitality sector, as well as increased panic-buying behaviour of consumers, thus leading to the iconic empty supermarket shelves seen in many countries around the world (Benton, 2020; Petetin, 2020; Power *et al.*, 2020). At the same time, high levels of on-farm food waste were observed in some agricultural sectors due to issues in redirecting supply to different outlets and labour shortages (Laborde *et al.*, 2020; Petetin, 2020; Wentworth, 2020; Yaffe-Bellany and Corkery, 2020). This highlighted a key lack of adaptability within the global food system, increasing calls for resiliency as shocks and stresses are expected to become only more common with climate change (FAO *et al.*, 2022).

1.1.4 Challenges facing the global food system

Considering the various aims of sustainable food systems, it can be seen that the main challenge is producing enough food of a suitable quality to ensure food security whilst also promoting human and environmental health. The FAO estimates that the agricultural sector will need to produce approximately 50% more food, livestock feed, and biofuel than in 2012 to feed a global population of 9.8 billion by 2050 (FAO, 2021; UN DESA, 2022). At the same time, agriculture depends upon finite natural resources such as land and water, which are being increasingly constrained, and this sector is also one of the most vulnerable to the pressing environmental, economic, and political issues of our time (Schmidhuber and Tubiello, 2007; FAO, 2021; Bezner Kerr, R., Hasegawa *et al.*, 2022; Farooq *et al.*, 2022). Indeed, although agriculture can be a major provider of ecosystem services, this sector is also known to contribute to a wide range of environmental impacts that threaten the long-term sustainability and resiliency of the food system (Horton, 2017). Thus, ensuring global food security within this set of challenges has been referred to as the 'perfect storm' of converging global issues (Beddington, 2010).

Today, agriculture covers approximately 43% of the world's ice- and desert-free land (Poore and Nemecek, 2018a), and irrigation accounts for 72% of freshwater withdrawals globally (FAO, 2021). As much of prime agricultural land is being lost to urbanisation, and global soils are increasingly being degraded by agricultural activity itself, the options to expand cultivated land to increase food production are limited (FAO, 2021). In addition, growing water scarcity, exacerbated by climate change, will further threaten the ability to increase food production (Falkenmark, 2013).

Indeed, climate change is one of the most prominent threats currently facing the agri-food sector. The changes in temperature and rainfall, increased pests and diseases, likely water shortages, rising sea levels, increasing soil salinity, and more frequent extreme weather events associated with climate change threaten the sector's ability to provide an adequate and stable supply of food for a growing global population (Schmidhuber and Tubiello, 2007; Vermeulen, Campbell and Ingram, 2012; Lamichhane *et al.*, 2015; Farooq *et al.*, 2022). These impacts are likely to exacerbate food insecurity, particularly in areas already vulnerable to hunger and undernutrition (Wheeler and von Braun, 2013). Even in countries that may not see marked crop yield declines initially, such as those with temperate climates, food access and utilisation will likely be indirectly impacted by changes in household income and damages to health (Schmidhuber and Tubiello, 2007; Wheeler and von Braun, 2013).

Many of the pressures on the world's land, water, and soil resources that are threatening future food production are actually being driven by the agri-food sector (FAO, 2021). Indeed, the global food system is known to be a major driver of climate change, contributing to approximately one-third of global anthropogenic greenhouse gas emissions, with the vast majority of these emissions (70-85%) coming from agricultural production and associated land use change such as deforestation (Vermeulen, Campbell and Ingram, 2012; Poore and Nemecek, 2018a; Mbow *et al.*, 2019; Crippa *et al.*, 2021). During agricultural production, major sources of emissions include: soil-based nitrous oxide (N_2O) emissions, mainly related to the use of nitrogen fertilisers; methane (CH_4) emissions from the fermentative digestion of ruminant livestock, flooded rice fields, and manure storage; and carbon dioxide (CO_2) emissions from the burning of plant litter, as well as the loss of organic carbon in croplands and rangelands due to land use change (especially deforestation) and intensive grazing (Smith *et al.*, 2007; IPCC, 2014). Additionally, the production of mineral nitrogen fertilisers requires considerable energy, and thus can be a major contributor of global warming impact for certain foods (Vermeulen, Campbell and Ingram, 2012); for example, this was seen to account for nearly 50% of global warming impacts from the production of a bread loaf (Goucher *et al.*, 2017). Finally, agriculture is also known to be a major polluter of soils and waterways, contributing to approximately 32% of terrestrial acidification and 78% of eutrophication impacts globally (Poore and Nemecek, 2018a).

The environmental impacts associated with agriculture are linked to the widespread utilisation of intensive farming practices that tend to be extractive rather than regenerative (Holt-Giménez and Altieri, 2013; Sánchez-Bayo and Wyckhuys, 2019). These practices include: the increasing use of chemical (synthetic) fertiliser inputs and pesticides; the employment of intensively cropped monocultures (single crop systems) with low genetic diversity; increased farm mechanisation; the use of high-yielding crop varieties that are often less resilient to pest and disease; and higher livestock grazing intensities, all of which are seen to contribute to land degradation and the pollution of surface and groundwater resources (IPBES, 2018; FAO, 2021). The increasing scale and intensification of agriculture became globally prominent beginning in the late 1960s, during what is known as the 'Green Revolution' (Jain, 2010). During this time, the U.S. and several European countries facilitated the uptake of high-yielding crop varieties, chemical fertilisers and pesticides, and increased mechanisation and irrigation across large parts of Asia and Latin America (Jain, 2010; John and Babu, 2021), in some cases replacing traditional, small-scale 'peasant' farming systems and subsistence agriculture (Harwood, 2011). As a result, these areas saw unprecedented agricultural growth, with yields of staple crops rising by anywhere from 1.5-3x during 1960-2010, with only a 30% increase in cultivated land area (Pingali, 2012; John and Babu, 2021).

Thus, the Green Revolution is often credited with preventing widespread famine and reducing hunger and poverty in many parts of the world (John and Babu, 2021). At the same time, this period resulted in an 800% increase in global fertiliser use and 100% increase in water resources used for irrigation since 1961 (Mbow *et al.*, 2019)**.** The increasing intensification, mechanisation, and dependence on external farm inputs and limited crop varieties now typifies what is termed as the 'industrial agriculture regime'; this has been cited as a main driver of current ecological degradation and the potential impoverisation of many small-scale farmers who were not able to make the transition, leaving some scholars questioning if this 'de-peasantisation' of the farming system has actually left many rural communities more vulnerable (Cullather, 2004; McMichael, 2009; van der Ploeg, 2010; Woodhouse, 2010; Harwood, 2011; Vanhaute, 2011; Holt-Giménez and Altieri, 2013; John and Babu, 2021).

Along with contributions to climate change, this type of intensive agriculture, now the norm in much of the world, has been cited as one of the main drivers of global soil degradation (FAO and ITPS, 2015; Lal, 2015). Soil degradation can be defined as the reduced capacity of soils to support life (i.e., loss of soil quality), characterised by erosion, compaction, acidification, salinisation, loss of organic matter, nutrient depletion, loss of soil biodiversity, pollution, and increased prevalence of resistant soil-borne pathogens, pests, and weeds (Chen *et al.*, 2002; Lal, 2015). Soil degradation may be caused by a variety of human-induced factors such as deforestation, overgrazing, agricultural mismanagement, overexploitation of vegetation, soil sealing, and bio-industrial activities (Bridges and Oldeman, 1999; Chen *et al.*, 2002; Bai *et al.*, 2008). In 2008, it was estimated that nearly 24% (3.5 billion ha) of the world's soil had been degraded, of which 19% of was cropland and 20-25% was rangeland (Bai *et al.*, 2008). More recent reports suggest that 75% of the world's land area is seeing negative impacts on ecosystem services and biodiversity as a result of land transformation and soil degradation (IPBES, 2018).

Soil degradation threatens the ability to produce food in the future, as well as compromising the many ecosystem services that flow from soil as a key natural capital asset (Gomiero, 2016). Degraded soils result in reduced resilience of agroecosystems, such that they are more negatively impacted by environmental stresses such as drought or soil diseases (D'Odorico *et al.*, 2013; Lehman *et al.*, 2015). By 2050, land degradation and climate change are predicted to reduce crop yields by an average of 10% globally and up to 50% in some areas (IPBES, 2018). These collective issues are predicted to result in 50 to 700 million people needing to migrate by 2050 (IPBES, 2018).

Agriculture has also been highlighted as a key driver of biodiversity loss due to associated land transformations such as deforestation, grassland conversion, and wetland drainage, as well as the homogenisation of farmed landscapes, which destroy important wildlife habitats (Benton, Vickery and Wilson, 2003; Kleijn *et al.*, 2009). Nutrient losses from farming systems can also lead to eutrophication, further threatening aquatic life (Withers and Haygarth, 2007; Withers *et al.*, 2014). Recent estimates suggest that 40% of terrestrial species loss has already occurred in the most intensively managed croplands globally, but even 20% loss has been estimated to have occurred in the most extensively managed pasture areas (Newbold *et al.*, 2015). In addition, it is predicted that a further 40% of the world's terrestrial and aquatic insect species may likely be extinct over the next few decades (Sánchez-Bayo and Wyckhuys, 2019). Intensive agricultural practices, including the planting of genetically-uniform monocultures, the prevalent use of synthetic fertilisers and pesticides, the removal of habitats such as hedgerows and trees, and the modification of soil surfaces to improve irrigation and drainage have all been linked to terrestrial and aquatic species decline, with these in particularly being highlighted as a main cause of insect species decline (Sánchez-Bayo and Wyckhuys, 2019). The loss of biodiversity threatens food security through the loss of important wild sources of food, pharmaceuticals, dyes, and other chemicals; additionally, biodiversity is vital to watershed management, soil health, erosion control, pollination, seed dispersal, nutrient cycling, and natural pest and disease control, all of which are important in both natural and farmed ecosystems (Muluneh, 2021).

It is thus clear that a range of environmental crises threaten future food production and thus food security; however, there are also socio-political factors that can impact the food supply. For example, the COVID-19 pandemic disrupted food supply chains and also intensified the barriers to food access and affordability, with an additional 150 million people across the world facing hunger since its outbreak, totalling up to 828 million worldwide in 2022 (FAO *et al.*, 2022). More recently, Russia's invasion of Ukraine in 2022 and the currently ongoing conflict is affecting the supply and trade of staple cereal crops, which has led to food shortages in some parts of the world, rising food prices, and a cost-of-living crisis, further threatening global food security (FAO *et al.*, 2022).

The current food system can thus be characterised by a state of severe imbalance that threatens food security (Garnett, 2014). Many across the world face malnutrition, characterised by the twin but opposing issues of undernutrition and obesity (Garnett, 2014; Steiner, Geissler and Schernhammer, 2019; FAO *et al.*, 2022). Additionally, although enough food is currently produced to feed the global population (Holt-Giménez *et al.*, 2012), and in fact this food supply is sufficient to feed the population in 2050 provided reductions in waste and changes in diet are achieved (Berners-Lee *et al.*, 2018), millions across the world still face hunger (FAO *et al.*, 2022). This is because the main problems of hunger and malnutrition are generally not associated with food supply, but rather affordability and access (Lang, 2010; Holt-Giménez *et al.*, 2012; Sage, 2013; Tomlinson, 2013). It is clear that any approaches toward global food security must therefore focus not just on increasing food production, but also ensuring that this food is available and accessible to all (De Schutter, 2014).

Considering the multi-faceted environmental, economic, and political challenges facing the world, increasing the sustainability and resiliency of the global food system has become a priority now more than ever (IPBES, 2018; Dimbleby, 2021; FAO, 2021; FAO *et al.*, 2022; IPCC, 2022). Current research purports a range of possible strategies in this regard, but consensus has not yet been reached (Garnett, 2014). The various approaches to sustainable and resilient food systems are explored in the following sections.

1.1.5 Approaches to sustainable and resilient food systems

Garnett (2014) defines three major perspectives on achieving food system sustainability (and thus food security), which are founded upon different basic worldviews. These include: efficiency-oriented; demand restraint; and food system transformation. The first focuses on new technologies and innovations to increase food production, assuming that these can be used to overcome or expand environmental limits. The second focuses on reducing excessive consumption through individual changes in consumer choice; this is based on a moral ground that humans are damaging and exceeding planetary limits, and thus must be limited themselves. The third approach is underlain by principles of social justice, seeing the main food system problems as structural and socio-economic; this therefore focuses on a need for structural systems change, rather than individual change as in the prior approach.

These basic perspectives are seen to pervade academic discourse on how food system sustainability should be achieved. In particular, these worldviews permeate some of the main approaches to sustainable agriculture, explored here as one (major) part of a sustainable food system and the main focus of this thesis.

1.1.5.1 Sustainable agriculture

Approaches to sustainable agriculture are often based upon two main strategies for managing land use: ‘land sharing’ and ‘land sparing.’ Acknowledging the limited amount of land available to expand agricultural production due to widespread soil degradation and increasing land competition for other uses (Oliver and Gregory, 2015), these strategies aim to meet the needs for future food production whilst also conserving soils and biodiversity (Green *et al.*, 2005; Phalan *et al.*, 2011). In the ‘land sharing’ approach, agriculture is blended within natural ecosystems to co-produce food along with a range of other ecosystem services; in other words, the land for food production is ‘shared’ with wildlife. An obvious example is agroforestry, which can involve growing crops in between rows of trees (Hardaker, Pagella and Rayment, 2021). On the other hand, the idea of land sparing is to increase food production per unit area so that theoretically, land elsewhere can be ‘spared’ from conversion to cropland and thus conserved for natural purposes.

Sitting at various points within the spectrum of land sparing to land sharing, Muller *et al*. (2017a) defines three main approaches to sustainable agriculture, which include: agroecological practices, intensification strategies, and high-tech, engineered approaches. Agroecological methods advocate the 'land sharing' approach, focused on meshing agriculture with natural ecosystem dynamics (Holt-Giménez and Altieri, 2013). On the farm, the focus is on building soil fertility; enhancing biological activity and above- and below-ground biodiversity; optimising nutrient use by recycling existing nutrients and biomass on the farm (e.g., manure and compost); closing energy and resource loops, and thus limiting the use of off-farm inputs; controlling pests and weeds through ecological management; and mitigating greenhouse gas emissions through carbon sequestration in the soil and reduced use of fossil fuels (Altieri, 1995; Altieri and Nicholls, 2018).

On the other end of the spectrum are engineering approaches, which focus on decoupling food production from the land and thus optimising control and efficiency of food production (Muller, Ferré, *et al.*, 2017). Examples include the production of lab-cultured meat and hydroponic (soilless) cultivation within a controlled environment. As these approaches generally do not depend on soil resources, they can be implemented on less viable land, such as in deserts or near cities; additionally, because generally higher yields are produced in controlled environment agriculture (Despommier, 2011; Barbosa *et al.*, 2015), this means that other land can be 'spared' for conservation purposes.

Also advocating the land sparing approach is the idea of 'sustainable intensification', although this generally requires higher land use per crop output than engineered approached and thus theoretically results in a lower amount of 'spared land' (Muller, Ferré, *et al.*, 2017). However, this approach is more applicable to 'mainstream', soil-based agriculture. The main idea of sustainable intensification is finding ways to maximise yields whilst minimising inputs and environmental impacts (The Royal Society, 2009). Examples include integrated pest management; precision agriculture technologies, which use global positioning systems (GPS) and remote sensing technologies to map crop growth and apply fertilisers, pesticides, and irrigation to more accurately match crop needs; and crop breeding or genetic modification to generate high-yielding and more robust crop varieties (Godfray *et al.*, 2010).

Of course, the viability of these approaches depends on local context and have various trade-offs. For example, engineered approaches are often limited by capital investment and maintenance costs, generally being more energy- and material-intensive (Muller, Ferré, *et al.*, 2017). At the same time, engineered approaches maximise land use per unit output; for example, hydroponic systems can be built vertically (e.g., stacking trays) to maximise aerial space and minimise the land footprint (Despommier, 2011; Muller, Ferré, *et al.*, 2017). Controlled environment agriculture also generally results in higher yields, as there are no limitations based on weather conditions, growing conditions can be optimised, and there are more growing cycles per year; however, these growing methods are often only suitable for certain fruit and vegetable crops (Despommier, 2011; Barbosa *et al.*, 2015).

There are also various environmental trade-offs when comparing land sparing and sharing strategies. Proponents of land sparing through sustainable intensification argue that greater biodiversity can be supported if larger areas of land are left as purely 'natural' or 'wild', rather than attempting to incorporate various 'wildlife-friendly' areas within an agricultural landscape (Green *et al.*, 2005; Phalan *et al.*, 2011). While land sharing models like agroecology may lead to greater biodiversity on the farm, the generally lower yields achieved from these systems mean that overall, more land is needed to produce a certain amount of food, thus reducing the land available for wildlife (Garnett, 2014). Further, the species supported by natural habitats on farms are different than those found on 'natural' or 'virgin' land and are potentially of less conservation interest (Phalan *et al.*, 2011; Garnett, 2014). However, there are also species that are best supported by wildlife-friendly farmed landscapes (e.g., hedgerows and field margins) and specific crop rotations, such as yellowhammers, hares, skylarks, and other bird species (Sausse *et al.*, 2015; Feniuk, Balmford and Green, 2019; Finch *et al.*, 2019; Dimbleby, 2021).

Another concern for land sparing is whether or not this occurs in actuality; indeed, Fischer *et al.*, (2011) highlights that the growth-based economic model of intensive farming may just result in the expansion of high-yielding production to increase profits further, thus undermining any theoretical land sparing benefits. Altogether, this has led to the support of a 'three compartment strategy', where intensive, high-yield farming, wildlife-friendly, lower-yield farming, and spared natural conservation areas are combined to provide a range of habitats to maximise biodiversity and ensure adequate food production (Feniuk, Balmford and Green, 2019; Finch *et al.*, 2019). Indeed, in Dimbleby (2021)'s National Food Strategy for England, a combination of land sparing and land sharing strategies is highlighted as providing the best outcomes for nature.

An additional consideration when comparing approaches to sustainable agriculture are the implications for the food system overall (Zurek *et al.*, 2018). Engineered, high-tech approaches and sustainable intensification follow the 'efficiency-oriented' mindset toward achieving food system sustainability, mainly focused on maximising production to achieve food security (Garnett, 2014). A crucial difference of agroecology is its additional focus on social justice (Bernard and Lux, 2017); thus, agroecology lends itself better to the 'food system transformation' perspective of food system sustainability, with the aim to create structural change that results in just, accessible, and resilient food systems (Garnett, 2014; FAO, 2018). A key focus is on ending hunger by increasing accessibility, rather than simply increasing productivity; other key ideals are the promotion of equity, the protection of farmers' livelihoods, and the conservation of natural resources (Wezel *et al.*, 2009; Holt-Giménez and Altieri, 2013; Fernandez *et al.*, 2018). Woven within the social justice aims of agroecology is the concept of 'food sovereignty'; this is a rights-based approach to food system design, where local people have the right to control their own food systems, food cultures, markets, natural resources, and modes of production (Whittman, 2011). Food sovereignty promotes the rights of local growers and smallholders over transnational corporations, encourages the use of regenerative and agroecological practices, and aims to secure the right to food for all (Pretty, Morrison and Hine, 2003; Windfuhr and Jonsén, 2005; Altieri, 2009; Altieri and Toledo, 2011; Vaarst *et al.*, 2018).

Agroecology has thus been described as a science, a set of practices, and a social movement all together (Wezel *et al.*, 2009), in which scientific and traditional smallholder knowledge is combined (Altieri, 1995; Altieri and Nicholls, 2018). Ecological and social principles are simultaneously applied in the design and management of food and agricultural systems, emphasising the use of diversified farming systems that replace synthetic external inputs (such as fertilisers and pesticides) with natural, on-farm processes (Wezel *et al.*, 2009; FAO, 2018). The concept of agroecology is not new, and indeed can be traced to the beginning of the 20th century; however, it is recently beginning to receive more attention by international organisations, such as the FAO, as a predominant approach to sustainable agriculture and achieving global food security (De Schutter, 2014; FAO, 2018, 2021).

1.1.5.2 Localising food systems

Shifting from the emphasis on food production, there has also been much academic focus on increasing the sustainability of the food supply chain overall, particularly in terms of increasing resiliency in the aftermath of the COVID-19 pandemic.

The COVID-19 pandemic was characterised by widespread and severe food insecurity, which spurred the need to rethink the food system to build resilience throughout (Garnett, Doherty and Heron, 2020; Hobbs, 2020; Bellamy *et al.*, 2021). During this time, global trade was disrupted, which impacted the ability to provide a consistent and stable supply of food to people (Béné, 2020; Petetin, 2020). In the UK, many factors influenced the instability of the food supply, including: a heavy reliance on food imports and specific international trade routes; a lack of supplier diversity for supermarkets, which resulted in deadlocked national supply chains; and reduced domestic food production due to labour shortages and other financial challenges (Garnett, Doherty and Heron, 2020; Laborde *et al.*, 2020; Petetin, 2020).

One strategy that has been commonly promoted to reduce dependence on complex and vulnerable global trade networks is increasing food self-sufficiency and shortening food supply chains, or otherwise building more local food systems (Béné, 2020; Fei, Ni and Santini, 2020; Garnett, Doherty and Heron, 2020; Hendrickson, 2020; Hickey and Unwin, 2020; Laborde *et al.*, 2020). This has been a key part of governmental food strategies in the UK (Defra, 2022c), the European Union (European Commission, 2020), and other countries around the world (Hickey and Unwin, 2020). Self-sufficiency in terms of food production refers to the extent that an area can satisfy the food needs of the population from domestic production (Clapp, 2017). On the other hand, shortening supply chains generally refers to reducing the intermediaries involved in the transfer of food from the farmer to consumer (European Commission, 2014). Although slightly different, both tend to be associated with local food systems (Stein and Santini, 2022), where “foods are produced, processed and retailed within a defined geographical area” (Kneafsey *et al.*, 2013); this can include, country, regional, and even community scales.

Increasing local food production can provide a buffer in times of global crisis, when international food supply chains prove ineffectual (Fei, Ni and Santini, 2020; Garnett, Doherty and Heron, 2020; Laborde *et al.*, 2020; Lal, 2020). The vulnerability of the global food system was seen with the COVID-19 pandemic, but also more recently with the rise in food prices in many countries as a result of Russia’s invasion of Ukraine (FAO *et al.*, 2022; Jagtap *et al.*, 2022; Mbah and Wasum, 2022; Alexander *et al.*, 2023). Breakdowns in the global food supply chain are only expected to increase with extreme weather events and political crises associated with climate change (Hsiang, Meng and Cane, 2011; Bowles, Butler and Morisetti, 2015; FAO *et al.*, 2022).

In the UK, a combination of domestic food production and imports from a diverse range of overseas suppliers is seen as critical to ensure food system resilience (Defra, 2021b). The UK supplies 60% of its own food by value and 54% by weight (Defra, 2021b). The majority of grains, meat, dairy, and eggs are produced within the country; however, for horticulture (fruits and vegetables), the UK is more reliant on imported produce due to the relatively colder climate and inability to grow many crops year-round (Defra, 2021b). In 2021, the UK imported 43% of fresh vegetables and 85% of fresh fruit by weight (Defra, 2022b). The UK is especially reliant on just two countries, Spain and the Netherlands, for its vegetable imports (Garnett, Doherty and Heron, 2020). Agricultural organisations, such as the National Farmers' Union and Soil Association (organic certification body), as well as environmental charities, such as Sustain and the Royal Society for the Protection of Birds, have all advocated the need to support local food production and build local food systems in the UK; this is highlighted as advantageous for local economies and for ensuring the production of better quality foods for both people and the environment (NFU, 2021a; Sustain and RSPB, 2021; Soil Association, 2022). Thus, finding innovative ways to increase sustainable, domestic food production in the UK, especially for fresh fruits and vegetables, are of particular interest for increasing food system resiliency.

Shortening food supply chains also brings the opportunity to place more power in the hands of farmers and consumers (Hendrickson, 2020), which has been a long-time goal of those advocating for agroecology, food justice, and food sovereignty (Altieri, 2009; Wezel *et al.*, 2009; Holt-Giménez and Altieri, 2013). This can help ensure that the food system serves the needs of the community – supporting the many and not the few (Sumner, 2011). As power is redistributed from the bottom-up, this can be seen to increase flexibility and adaptability in food systems (Hendrickson, 2020); indeed, one of the main issues during COVID-19 was the inflexibility built into national food supply chains, which prevented farmers from re-directing supplies from the hospitality sector to meet increased demand in retail environments (Garnett, Doherty and Heron, 2020; Wentworth, 2020).

On a community-scale, various types of local agriculture, including local farms, community gardens and allotments, as well as local shops, played an important role in helping people access food during the pandemic (Busby, 2020; Hobbs, 2020; Schoen *et al.*, 2021; Tittonell *et al.*, 2021; DuPuis, Ransom and Worosz, 2022; Jones, Krzywoszynska and Damian Maye, 2022). However, it is unclear whether the role of local agriculture was widespread or just concentrated to specific reported case studies (DuPuis, Ransom and Worosz, 2022). Additionally, although the social and community benefits of localising food production and shortening food supply chains is generally agreed upon within academic literature, the environmental benefits are less certain (Chiffoleau and Dourian, 2020; Stein and Santini, 2022). Smaller-scale, localised food systems have been highlighted as being less cost-efficient and offering less variety for consumers (Hobbs, 2020), which may result in undesired dietary and lifestyle changes (Hickey and Unwin, 2020). Stein and Santini (2022) have thus cautioned policy-makers from blindly supporting increased food self-sufficiency or local agriculture. However, types of hyper-local food production (such as home-growing, community gardens, and local farms) have been consistently shown to improve resiliency to crises in the past (Hamilton *et al.*, 2014; Mok *et al.*, 2014). With an estimated 68% of the global population living in urban areas by 2050 (UN DESA, 2019), this has also led to calls for increased food production within cities to further build resiliency (Kennard and Bamford, 2020).

1.1.5.2.1 Urban agriculture

The FAO (2007) defines urban agriculture (UA) as "the growing of plants and the raising of animals for food and other uses within and around cities and towns, and related activities such as the production and delivery of inputs, processing and marketing of products." Mougeot (2005) further specifies that UA is integrated into the local economic and ecological system of cities and can also include nearby towns and suburbs (peri-urban areas) that supply to urban areas. Peri-urban agriculture (PUA) generally refers to agricultural activities taking place at the fringes of cities, in the transition zone between urban and rural areas, characterised by lower populations densities and less infrastructure than urban areas (Opitz *et al.*, 2016). The boundaries used to specify urban versus peri-urban agriculture depend on the particular region and purpose of study. There are also different scales of UA, which can largely be categorised into personal (e.g., home gardens), community (e.g., community gardens, school gardens, and allotments), and commercial (e.g., small-scale farms and indoor farms) (Kennard and Bamford, 2020).

Globally, the potential for UA to contribute to food production appears to be low; a recent mapping study predicted that, with maximum production and space use, UA could produce 5% of the total agricultural production of pulses, roots and tubers, and vegetables, although likely only contributes to 1% of this production currently (Clinton *et al.*, 2018). However, on the city-scale, UA can significantly contribute to food self-sufficiency. For example, Shanghai, China produces 50% of vegetable demand within the city (Lang and Miao, 2013), and the city of Sydney, Australia is able to produce 24% of the state's total production of vegetables (Mok *et al.*, 2014). In Sheffield, UK, a recent mapping study showed that the available green space for food growing would equate to more than four times the current per capita footprint of commercial horticulture in the UK (Edmondson *et al.*, 2020). If urban horticulture growing was practised in just 10% of domestic gardens and 10% of additional green space, this could produce enough fruits and vegetables to feed 15% of the Sheffield population per year (including current allotment production) (Edmondson *et al.*, 2020). The study also highlighted that controlled environment agriculture (CEA) on rooftops could further contribute to the production of high-value fruits and vegetables (Edmondson *et al.*, 2020).

In the past, political and economic crises have led to increases in both home growing and urban food production (Hamilton *et al.*, 2014; Mok *et al.*, 2014). For example, during the World War II *Dig for Victory* campaign, allotments in the UK provided about 10% of the nation's food by weight and about half of its fruit and vegetables (Crouch and Ward, 2003). Similar 'victory gardens' were also seen in the U.S. at this time (Bassett, 1981). In Cuba, the fall of the Soviet Union in 1989 led to an economic crisis that spurred the growth of UA (Altieri *et al.*, 1999; Koont, 2008; Fernandez *et al.*, 2018). In 2014, 560 km^2 of urban and peri-urban agriculture sites in Cuba produced over 50% of all fresh produce for the country (Hamilton *et al.*, 2014; Companioni, Rodríguez-Nodals and Sardiñas, 2016; Altieri and Nicholls, 2018). The government was crucial to the success of UA in Cuba, providing training and technical advisory services (i.e., 'extension services') to new farmers in cities, increasing land access rights of individuals and groups to grow on state and vacant land, and supporting local markets and worker cooperatives (Koont, 2008). More recently during the COVID-19 pandemic, preliminary research suggests that home growing may have also increased in some towns and cities to supplement food supply, potentially leading to an increased protective effect over perceived food security and wellbeing in the early stages of the pandemic (Mead *et al.*, 2021; Mullins *et al.*, 2021).

Thus, UA and PUA have long been espoused as ways to contribute to local food security and therefore resiliency in cities by increasing access to fresh and healthy foods, especially in food insecure areas (FAO, 2007; Zezza and Tasciotti, 2010; McClintock, Cooper and Khandeshi, 2013; Hamilton *et al.*, 2014; Mok *et al.*, 2014; Saha and Eckelman, 2017). Integrating agricultural spaces within the built environment can also provide a wide range of supporting, provisioning, and cultural ecosystem services for cities, as well as leveraging synergies for other urban problems such as waste and water cycling (Pearson, Pearson and Pearson, 2010; Lovell and Taylor, 2013; Ackerman *et al.*, 2014; Proksch, 2017). For example, UA spaces in cities can be important for reducing local air temperatures, improving air and water quality, intercepting stormwater, remediating soils, sequestering carbon, providing a refuge for pollinators, and enhancing biodiversity via habitat creation for wildlife (Oberndorfer *et al.*, 2007; Pearson, Pearson and Pearson, 2010; Beniston and Lal, 2012; Lovell and Taylor, 2013; Brenda B Lin, Philpott and Jha, 2015; Goldstein *et al.*, 2016; Clinton *et al.*, 2018). UA sites can also provide an interactive space for individuals living in urban areas to reconnect to their food system and the natural environment, improving personal health, fostering feelings of relaxation and well-being, and creating opportunities for community engagement and education (Tzoulas *et al.*, 2007; Wakefield *et al.*, 2007; Pearson, Pearson and Pearson, 2010; Turner, 2011). Thus, while UA may not be a viable solution to provide for all food needs of urban residents, shifting some food production to cities can help reduce pressure on current agricultural land while also providing other environmental and social benefits to the urban landscape (Clinton *et al.*, 2018; Wilhelm and Smith, 2018).

However, the sustainability of farming within an urban context is uncertain. For one, it is often difficult to find available space within densely populated urban areas, and finding land that is suitable for farming is even harder (Daftary-Steel, Herrera and Porter, 2015; Siegner, Sowerwine and Acey, 2018). Urban soils can be variable and of poor quality due to compaction and degradation, which may affect crop yields (Beniston, Lal and Mercer, 2016; Lal, 2020). Additionally, cities are sources of air, water, and soil pollution, including possible heavy metal contamination from industrial activity and urbanisation, which leads to concerns about food safety (Säumel *et al.*, 2012; Mitchell *et al.*, 2014; Antisari *et al.*, 2015; Balotin *et al.*, 2020). Urban farmers may also lack access to small-scale agricultural equipment, opportunities for bulk purchasing of supplies, and educational opportunities, such as relevant extension (advisory) services (in the U.S.), which altogether may make it more difficult for urban farms to remain economically viable in the long-term (Kennard and Bamford, 2020). These combined challenges of soil quality and lack of appropriate services for small-scale farms may lead to higher resource use (McDougall, Kristiansen and Rader, 2018), possibly negating any environmental benefits from shorter food transport distances.

Thus, when evaluating the potential to increase local food production and self-sufficiency in a sustainable manner, it is necessary to compare different models of local production and their environmental trade-offs, which will ultimately be affected by the use of various technologies, farmers' management practices, varying soil quality in different areas, yields, and farmer experience (Edwards-Jones 2010). By considering these trade-offs in a local context, this can help to inform national and city policy-makers so that they can promote strategies to build local food systems specific to the needs and challenges of their population and environment. Chapter 3 of this thesis thus explores trade-offs between different modes of local agriculture, and different agricultural management types, within the U.S. and UK context using lifecycle assessment. The following Section 1.2 provides further background specific to this study.

1.2 Evaluating environmental impacts from local agriculture using lifecycle assessment

As a key part of food system sustainability, a wide array of research has focused on finding ways to reduce the environmental impacts associated with food production (Green *et al.*, 2005; Foley *et al.*, 2011; Garnett, 2011; Muller, Ferré, *et al.*, 2017; Muller, Schader, *et al.*, 2017; IPCC, 2022), as discussed in Section 1.1.5. Two areas that have been focused on widely, both in literature and in the public scope, are 1) employing more environmentally-friendly farming practices, such as organic agriculture, and 2) reducing food system impacts by shortening food supply chains. A key way to assess the environmental impacts of these strategies is through lifecycle assessment; although perhaps lending itself to efficiency-oriented approaches (Garnett, 2014), this method can still provide considerable insight to environmental impact 'hotspots' in specific parts of the food system and key trade-offs between sustainable agriculture approaches (Horton *et al.*, 2017). This introductory section thus provides context for the evaluation of 'organic' and 'more local' food production as potential sustainable agriculture strategies through lifecycle assessment.

1.2.1 Organic agriculture

Organic agriculture is built on many of the same principles and practices as agroecology, and thus is often seen as a form of agroecology (Wezel *et al.*, 2014), with its main aim to improve and care for ecosystem, human, and animal health (FAO, 1999; IFOAM, 2020). However, organic agriculture perhaps gains more attention in research and in the public eye due to the rigorous and well-known organic certification and food labelling programmes (Janssen and Hamm, 2012; Migliorini and Wezel, 2017; IFOAM, 2019). Organic agriculture can generally be defined as a holistic farming system that enhances ecosystem health by promoting biodiversity, building soil life, and embedding food production into natural and biological cycles (FAO, 1999). Organic systems focus on ecosystem management, using ecological processes and local resources to create closed nutrient cycles on farms rather than relying on external, off-farm inputs (IFOAM, 2020). The focus is on using management decisions to build long-term soil fertility and to prevent pests and disease. Generally, this means that synthetic fertilisers and pesticides, veterinary drugs, genetically modified seeds, and certain food additives are restricted from use on organic farms (IFOAM, 2020).

Internationally, organic farming is regulated through the Codex Alimentarius Guidelines (FAO & WHO, 2007), established by the FAO and World Health Organization (WHO), as well as through the International Federation of Organic Agriculture Movements (IFOAM) standards (IFOAM, 2019). However, specific regulations regarding organic certification are often designated on a national level and verified by independent third-party reviewers. These regulations define the types of inputs (e.g., fertilisers and pesticides) or veterinary drugs allowed for use on farms aiming to hold organic certification and also provide a holistic overview of how management systems should be operated (e.g., using crop rotations, building soil health). Thus, specific allowances for organic agriculture and the bodies awarding certification will vary by country (Esteves, Vendramini and Accioly, 2021).

Additionally, many small farms may forego organic certification, even if following organic guidelines, due to certification costs (Esteves, Vendramini and Accioly, 2021). This has led to the creation of other certification programmes that still follow basic organic guidelines, for example the farmer-to-farmer 'Certified Naturally Grown' certification programme in the U.S. (https://naturallygrown.org/).

Organic foods are often viewed by consumers to be healthier, of better quality, and more environmentally sustainable (Schifferstein and Oude Ophuis, 1998; Shafie and Rennie, 2012; Siegrist, Visschers and Hartmann, 2015), and these topics are also widely portrayed in media and discussed in online platforms, mainly in a positive framing (Cahill, Morley and Powell, 2010; Danner *et al.*, 2022; Diaconeasa *et al.*, 2022). Organic agriculture is seen to contribute to climate change mitigation through the reduced emissions associated with the absence of synthetic fertiliser use and enhanced carbon sequestration potential (Scialabba and Müller-Lindenlauf, 2010; FAO, 2011), as generally higher levels of soil carbon and organic matter are found in organically-farmed soils (Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012). In addition, many of the typical practices used in organic farming systems, such as maintaining soil cover, utilising diverse crop rotations that incorporate legumes, and increased application of organic fertilisers to soil, have been highlighted by the Intergovernmental Panel on Climate Change (IPCC) as contributing to climate change mitigation through soil carbon sequestration (IPCC, 2022).

However, there is a concern about whether or not there is enough evidence to support these views for organic agriculture, especially as there are many trade-offs also commonly associated with the practice. Organic agriculture is generally seen to result in yields that are 75-80% of those achieved in conventional farming systems (De Ponti, Rijk and Van Ittersum, 2012; Seufert, Ramankutty and Foley, 2012). Thus, one of the main drawbacks to employing organic agriculture globally is that it may come at a significant land cost (Muller, Schader, *et al.*, 2017), which could actually lead to higher greenhouse gas emissions through land use change (IPCC, 2022). However, organic agriculture has also been seen to result in a number of positive ecosystem benefits in certain instances, such as improved soil health and biodiversity (Maeder *et al.*, 2002; Bengtsson, Ahnstrom and Weibull, 2005; Tuomisto *et al.*, 2012), although again there is still academic uncertainty about the scope of these benefits (Hole *et al.*, 2005; Lorenz and Lal, 2016).

1.2.2 Local agriculture

Local food production, which is generally defined as food grown and consumed within a set geographic region, is perceived to have several benefits, including improved food quality and safety, as well as reduced environmental impacts (Feldmann and Hamm, 2015; Siegrist, Visschers and Hartmann, 2015; Stein and Santini, 2022). These reduced impacts are commonly portrayed in terms of lower food miles, or the distance that food travels from farm to consumer (Kemp *et al.*, 2010). The international food supply chain relies on extensive global trade networks (Aday and Aday, 2020; Li *et al.*, 2022). Even within a country, food can travel extensive distances from farm to market; for example, fresh produce may travel >2000 km from farm to market across the east and west coast of the U.S. and Canada (Dyer *et al.*, 2011).

As the world population is becoming increasingly more urbanised (Satterthwaite, McGranahan and Tacoli, 2010; UN DESA, 2019), the transport of food has become one of the largest unidirectional flows into cities (Rufí-Salís *et al.*, 2021). Thus, the idea of 'hyper-local' food production has also gained attention, especially in reference to urban and peri-urban agriculture (Garcia, 2019; Henley, 2020); indeed, the U.S. Department of Agriculture has recently begun grant programmes to expand various types of food growing efforts in cities, including commercial farms, community gardens, and school projects, among others (USDA, 2022; USDA Press, 2022). Producing food in and around cities is seen to have the benefit of shortening supply chains, whilst also creating opportunities for synergies with other urban flows, for example in terms of waste cycling, wastewater treatments, and rainwater harvesting (Goldstein *et al.*, 2016).

While one of the main environmental benefits of increasing local food production is to reduce food miles, many studies have shown that transport impacts do not play a large role in food supply chains, especially when observed on a global or national level (e.g., through national dietary consumption patterns). For example, Weber and Matthews (2008) showed that final transport of food from producer to retail contributes to only 4% of the average U.S. household's carbon footprint for food consumption. Further, Benis and Ferrão (2017) found that localising all food production to occur within the Lisbon metropolitan area (average transport of 30 km) lowered the greenhouse gas emissions associated with the Portuguese diet by only 2-5%. Additionally, in some cases importing foods may result in lower greenhouse gas emissions, if producing them locally requires more energy-intensive methods; for example, this is commonly seen in the case of imported Spanish or Italian tomatoes, grown in fields or polytunnels, which often result in lower impacts than producing them in heated glasshouses in countries with colder climates, like the UK (Williams *et al.*, 2009; Webb *et al.*, 2013; Theurl *et al.*, 2014; Frankowska, Jeswani and Azapagic, 2019).

However, a recent study has brought new attention to the contribution of transport to food system emissions (Li *et al.*, 2022). The study found that food transport emissions contribute to approximately 20% of total food system emissions when accounting for the entire upstream food supply chain and including both international and domestic transport distances, which is 3.5-7.5x higher than previous estimates (Li *et al.*, 2022). The contribution from transport is found to be especially high for fruit and vegetable consumption, where global freight transport of these foods contributes to almost twice the amount of emissions released during their production. However, the study also found that if all people only consumed domestically-produced foods (i.e., foods produced in the same country), this would reduce emissions associated with food miles by only 9%, as road transport features a higher emission intensity per weight than maritime shipping.

For this reason, other studies have also shown that transport can play a more significant role in food impacts when considered on a local scale, for example, when comparing different regions of production (Meisterling, Samaras and Schweizer, 2009; Rothwell *et al.*, 2016; Hu *et al.*, 2019). Additionally, crops that are transported by air freight may also see higher impact contributions from transport, with Frankowska, Jeswani and Azapagic (2019) finding that air-freighted fresh vegetables had global warming potentials approximately five times higher than home-produced vegetables in the UK, for crops grown without energy-intensive methods (e.g., not tomatoes). Altogether, this shows that the role of food transport in food supply chain emissions may be higher than previously recognised and may be particularly important on a local level. This highlights a potential role of other forms of hyperlocal production, like urban and peri-urban agriculture, to reduce environmental impacts associated with the food supply chain.

However, despite the many perceived benefits of local and organic agriculture, it is clear that there is still academic uncertainty regarding the environmental benefits and trade-offs of these approaches. As various researchers have recently argued for more localised, small-scale, and/or agroecological food production in and around cities (Altieri and Nicholls, 2018; Lal, 2020; Langemeyer *et al*., 2021), this begs a need for more quantitative and objective assessments of the impact trade-offs between different forms of local production and the different types of management practices that may be utilised. Evaluating this within local geographical contexts would allow for specific recommendations to be made regarding ideal local agricultural models for a particular city or state.

1.2.3 Lifecycle assessment (LCA)

One of the main methodologies used to explore and compare the environmental impacts between different farming systems and food supply chains is lifecycle assessment (LCA). Lifecycle assessment involves evaluating the environmental impacts of a product or process over its entire lifecycle, including raw material extraction, production, use, and disposal (Muralikrishna and Manickam, 2017). Material, energy, and resource inputs like water and land use are quantified, and environmental impacts are calculated using robust models that relate these resource flows to specific emission outputs (Bjørn *et al.*, 2017). LCAs can cover a broad range of environmental impacts, such as global warming, eutrophication, acidification, and eco-toxicity, among others, thus expanding from a simple carbon footprint (Bjørn *et al.*, 2017). Therefore, LCAs allow for trade-offs and 'problem shifting' to be explored between different lifecycle stages, geographies, or environmental issues (Finnveden *et al.*, 2009). For these reasons, LCA has become a widely used methodology for analysing potential environmental impacts and comparing production choices or different products, thus informing decision-making in both business and policy (Henryson *et al.*, 2020).

LCA was originally developed for the analysis of industrial systems (Heijungs *et al.*, 1992), first being used in the 1970s to compare impacts between glass and plastic bottles by Coca-Cola (Baumann and Tillman, 2004). In the past two decades, LCA has also been used to assess environmental impacts from agriculture (van der Werf, Knudsen and Cederberg, 2020). This method now forms the basis of the European Union (EU)'s methodology for calculating environmental footprints of products, which includes certain food items (European Commission, 2022).

The stages of conducting an LCA follow a standardised framework as set out by the International Organization for Standardization (ISO) in ISO 14044 (ISO, 2006b). This includes first defining the goal and scope of the LCA, including its purpose, the system to be examined, and the methods used to calculate impacts. Resource, energy, and material flows as well as relevant emissions within the lifecycle of interest are quantified and modelled through lifecycle inventories (LCI). During the lifecycle impact assessment stage (LCIA), these flows and emissions are then linked to specific environmental impact categories (classification), and burdens are calculated using the characterisation factors defined within impact assessment methodologies (characterisation). The LCA practitioner then discusses and interprets these results within the final stage.

Although LCA is a powerful tool to assess a wide range of environmental impacts, many challenges and uncertainties still exist in its use (Henryson *et al.*, 2020). As a method, LCA is intended to estimate environmental impacts from systems, and thus does not necessarily portray actual emissions within a local context. LCA relies on many assumptions, both within the emission models and impact assessment methods used (Reap *et al.*, 2008; Schrijvers, Loubet and Sonnemann, 2016), and also in terms of what processes are considered and how these are allocated to the item of interest. The LCA practitioner decides where to draw the system boundaries for the LCA, and thus what items and processes to include; ideally, lifecycles would be assessed from 'cradle-to-grave', or from raw material extraction through end of life and waste disposal, but this is not always possible or necessary depending on the information available and the goal of the LCA. Thus, 'cradle-to-gate' or 'gate-to-gate' methods may be used, where only a portion of the product lifecycle is considered. For example, in many agricultural LCAs, only cultivation or 'on-farm' processes will be evaluated; if farming systems are being compared that also utilise different supply chains, then important differences may be lost in this case.

The methods of allocating burdens from multifunctional systems is one of the most hotly contested topics within LCA methodology (Ardente and Cellura, 2012). LCA is commonly used to identify impacts from the lifecycle of a singular product, but often a specific process may produce several products at the same time ('co-production'), or there may be many overlapping lifecycles for a product (i.e., 'cascading' systems), for example in the case of recycling (Tillman *et al.*, 1994; Ekvall and Tillman, 1997). ISO 14044 standards recommend first trying to avoid allocation entirely, either by dividing processes into concrete sub-processes that can be evaluated separately, or by expanding the system to include all co-products, functions, and lifecycles (ISO, 2006b). However, often these options are not possible, as detailed information may not be available for all stages of the lifecycle or all co-products. In this case, allocation becomes necessary, and while the ISO standards delineate the order of preferred methods, it is ultimately up to the LCA practitioner to decide how to allocate burdens. As LCA results can be extremely sensitive to how burdens are allocated between products, it is important for LCA practitioners to clearly state how allocation has occurred and, ideally, provide results for a number of allocation scenarios when this is found to influence results (Ardente and Cellura, 2012), as recommended in ISO methodology. It is clear that LCA results can be influenced by bias or subjectivity of the LCA practitioner, so caution should be exercised whenever using these results to inform decision-making.

Specific challenges also exist for agricultural LCAs in particular. A major component of LCAs on food products is the modelling of soil-based agricultural emissions, such as nitrous oxide (N_2O), ammonia (NH_3), and nitrogen oxide (NO_x) emissions to air, which generally occur as a result of fertiliser application and organic N mineralisation, as well as nitrate (NO_3^-) and phosphorus (P) leaching and runoff. Impact categories with increasing political relevance, such as climate change and marine eutrophication, are often dominated by these types of emissions (Nemecek *et al.*, 2016; Henryson *et al.*, 2019). These emissions are generally modelled in LCAs using generic emission factors such as those developed by the IPCC (IPCC, 2006c, 2019), which are applied to the total amount of nitrogen input in a linear fashion. However, geographical location and site-dependent factors can play a large role in these emissions, and thus models that do not account for spatial variability may present misleading results (Henryson *et al.*, 2020). Although there is a clear need for more site-dependent emission models, not many are available, and those that are come at a trade-off with the amount of time and detail required to use them (Avadí *et al.*, 2022).

Despite the many challenges associated with LCA, it is still one of the best methods available to assess and compare environmental impacts between different systems. The use of LCA in food systems has provided critical insight to impact hotspots in farming systems and food supply chains and the impact trade-offs that might erupt between alternative production methods. The following section summarises a selection of LCA studies that are relevant to this research, which assess the potential role of organic agriculture and local agriculture in reducing environmental impacts, specifically in the context of fruit and vegetable production.

1.2.3.1 LCAs on organic agriculture

There is a wide range of LCA literature comparing the use of organic vs. conventional production, both for livestock as well as cropping systems (Tuomisto *et al.*, 2012; Perrin, Basset-Mens and Gabrielle, 2014; Meier *et al.*, 2015). Although studies tend to show high variability, by analysing the literature collectively this has allowed for general conclusions to be drawn about the impact trade-offs between these systems.

It is generally accepted that conventional agriculture results in lower impacts per unit crop output, and organic agriculture has lower impacts when evaluated per unit area (Tuomisto *et al.*, 2012; Meier *et al.*, 2015). This is because organic agriculture usually has lower inputs per unit area, but also lower yields (Tuomisto *et al.*, 2012; Meier *et al.*, 2015). Despite this widely-held assumption, certain reviews have found that in many cases impacts between organic and conventional cropping systems are actually fairly similar (Nemecek *et al.*, 2011; Tuomisto *et al.*, 2012). Indeed, Tuomisto *et al.* (2012) reviewed a wide range of field-based, modelling, and LCA studies comparing environmental impacts and other environmental indicators between conventional and organic farming systems in Europe. The review found significant differences between systems only for energy and land use out of the evaluated LCA impact categories; in this case, organic systems were seen to have lower energy requirements and emissions per unit area, but higher land use.

The majority of LCA studies comparing organic and conventional cropping systems tend to focus on primary energy and global warming potential as the main environmental impacts for comparison (Meier *et al.*, 2015), with fewer studies also evaluating impacts such as eutrophication, acidification, and toxicity. Tuomisto *et al.* (2012)'s meta-analysis showed that organic cropping systems tended to have lower nutrient losses per unit area, but higher per product unit in comparison to conventional agriculture, which resulted in organic farms tending to have higher eutrophication and acidification potentials per product unit. However, the literature varies in these outcomes, and Nemecek *et al.*, (2011) proposes that higher nutrient losses for organic agriculture are usually seen in cases of high manure use. In terms of toxicity, organic production has been seen to result in lower toxicity impacts per crop produced, due to the absence of pesticide use (De Backer *et al.*, 2009; Nemecek *et al.*, 2011). This implies that there are impact trade-offs between conventional and organic agriculture that need to be further explored.

Indeed, many studies have criticised the use of LCA to compare organic and conventional agriculture, highlighting that the methodology itself tends to favour conventional agriculture and that many of the benefits of organic agriculture are lost (Meier *et al.*, 2015; Boone *et al.*, 2019; van der Werf, Knudsen and Cederberg, 2020). This is because LCA focuses on quantifying environmental impacts, or the negative outcomes of production, in terms of product efficiency. Thus, this favours conventional approaches to sustainable agriculture, such as sustainable intensification, where the aim is to maximise crop output whilst minimising negative impacts (The Royal Society, 2009; Godfray *et al.*, 2010). However, the goal of organic agriculture is to promote and enhance ecosystem health, not simply reduce impacts (FAO, 1999; IFOAM, 2020). Thus, many of the steps taken to improve ecosystem health, for example in cases of soil health and biodiversity, are often not accounted for.

In some cases, carbon sequestration potential of soils or from specific processes such as compost application have been considered in agricultural LCAs (Meisterling, Samaras and Schweizer, 2009; Venkat, 2012; Saer *et al.*, 2013). However, there is no standardised way to account for this, resulting in wide variation in the values used and how this is applied within LCAs. Meisterling, Samaras and Schweizer (2009) explored the application of a wide range of carbon storage values for both organic and conventional systems and found that global warming potentials were highly sensitive to the levels chosen. Venkat (2012) considered carbon sequestration potential mainly in relation to organic agriculture, modelling the additional carbon that can be sequestered on farms transitioning from conventional to organic production; this resulted in transitioning systems having approximately 18% lower global warming emissions than conventional production and 22% lower than steady-state organic systems, when evaluated across twelve fruit, vegetable, and nut crops.

Soil quality and biodiversity indicator scores have also been developed for use within LCA (Oberholzer *et al.*, 2006; Jeanneret *et al.*, 2014), but these are not widely applied (Meier *et al.*, 2015). Nemecek et al. (2011) utilised these indicator scores within a comparison of integrated organic and conventional farming systems in Switzerland and found that biodiversity scores were higher on organic farms, mainly due to the absence of pesticide use, but that soil quality scores did not largely differ. Other studies have aimed to develop new methods to account for the ecosystem services of farming systems, for example by allocating impacts between crop and ecosystem services outputs (Boone *et al.*, 2019), but again the application and development of these methods are still in their early stages. Thus, while LCA can be a powerful tool to compare environmental impacts from cropping systems, it is important to recognise the limitations of this methodology and discuss the additional trade-offs that might be lost within LCA frameworks.

1.2.3.2 LCAs on urban and peri-urban agriculture

Within the scope of local agriculture, LCA is typically used to compare domestically-produced foods to imported foods (Blanke and Burdick, 2005; Webb *et al.*, 2013; Theurl *et al.*, 2014; Payen, Basset-Mens and Perret, 2015). Fewer studies have compared different types of local agriculture within one region, outside of organic and conventional comparisons. However, in recent years there has been a growing body of literature evaluating urban and peri-urban agriculture through LCA. The majority of these studies come from the Universitat Autònoma de Barcelona (Autonomous University of Barcelona) (Dorr *et al.*, 2021) and focus on soilless or greenhouse cultivation in rooftops (e.g., Sanyé-Mengual *et al.*, 2015; Boneta *et al.*, 2019; Rufí-Salís *et al.*, 2020, 2021). There have been markedly fewer studies evaluating and comparing different scales of soil-based production, which are more relevant to this research.

Kulak, Graves and Chatterton (2013) provided what is often deemed as the first LCA study on urban agriculture (Dorr *et al.*, 2021). This study evaluated the global warming impacts from the production of twelve fruit and vegetable crops on a community farm situated on the urban fringes of London, UK, comparing this with impacts from the same crops as sold through typical British supermarkets (Audsley *et al.*, 2009). Impacts were considered from crop cultivation through to the retail phase. Results showed that most crops produced by the community farm resulted in lower impacts than their counterparts in the conventional food supply chain, seen for all crops except for strawberries produced in polytunnels. The greatest differences in impacts between systems were generally observed for crops that were conventionally produced using energy-intensive methods (e.g., UK tomato production in heated greenhouses) or that were imported into the UK from outside Europe via air freight. However, despite being one of the most widely cited LCA studies on urban agriculture (Dorr *et al.*, 2021), it relied extensively on literature data from other sources, even for the community farm case study. For example, yields for the community farm were not measured, but mainly based on those reported for organic crops at the University of West Virginia's research farm (Childers, 2005). Additionally, global warming potentials from on-farm cultivation activities were derived from literature; in some cases, this came from UK sources (Williams, Audsley and Sandars, 2006; Williams *et al.*, 2009), but not always. The large dependence of this study on secondary datasets implies the need for more primary data regarding specific yields and production practices used on urban and peri-urban farms.

In a later study, Rothwell *et al.* (2016) evaluated the influence of locality and the use of soilless cultivation on the environmental impacts of lettuce sold in Sydney, Australia's central vegetable market. Five case study farms were evaluated, including one larger-scale, interstate field-based operation in Victoria (930 km to market), and four smaller-scale peri-urban farms within the Sydney basin (<60 km to market), of which: two utilised field-based production; one used low-tech hydroponic methods; and one used high-tech hydroponic methods. Overall, the lowest global warming potentials were observed on the peri-urban farms with field-based operations, although these farms were seen to exhibit higher water consumption and land use in comparison to the hydroponic farms. The larger-scale interstate farm actually had lower global warming potentials for on-farm operations, but this was negated when post-harvest processing, storage, and transport burdens were included. Interestingly, this study showed that packaging and transport to market constituted the majority of global warming impacts for the larger-scale interstate farm, instead of on-farm processes, which are generally seen to dominate climate change impacts for conventional vegetable production (Perrin, Basset-Mens and Gabrielle, 2014; Poore and Nemecek, 2018a). However, on-farm processes dominated impacts for the peri-urban farms, as also seen in other LCAs on urban and peri-urban agriculture (Dorr *et al.*, 2021).

More recently, Hu *et al.* (2019) used LCA to compare global warming impacts between two urban farms in Beijing, China: one being a small conventional household farm that sold vegetables to consumers in a local market, and the other a larger urban farm that delivered produce directly to consumers homes and utilised mainly organic practices. Global warming impacts for the latter case were approximately double that of the prior. As these two farms were both characterised as 'urban' and were generally located close to the final consumer (<50km), impact differences were seen to be driven more by production practices and yields rather than transport. Although the larger farm used no mineral fertiliser or pesticides, unlike the small conventional farm, the higher yield of the conventional farm offset the burdens of these inputs, resulting in the lower global warming potential. However, this study also compared global warming impacts for both farms per unit of profit; for this case, the larger, organic farm had a slightly lower global warming impact than the conventional farm, due to the higher price point achieved for producing in a more sustainable or 'green' manner. This study also found that global warming impacts from these urban farms were higher than that reported as the national average for vegetable production in China (Yue *et al.*, 2017), which was mainly due to the more intensive nature of urban production (e.g., using greenhouses).

As can be seen, the main studies on local or urban agriculture that exist tend to focus on evaluating a small subset of case study farms (typically <5). Some studies, such as Kulak, Graves and Chatterton, (2013), compared local produce to what might have been otherwise purchased in a supermarket, including both home-produced and imported foods; other studies, like Rothwell *et al.* (2016), focused on comparing different levels of regional or local production. However, to the author's knowledge, few studies exist that compare conventional, large-scale production within one country to different modes of 'hyper-local' production (e.g., urban and peri-urban) at the same time. As LCA research on urban and peri-urban agriculture is still in its early stages, it is clear that more evidence is needed before drawing any wide-ranging conclusions about the sustainability of different types of local agricultural models.

1.3 Soil health and ecosystem services from local farms in the UK

Lifecycle assessments are an essential tool to approximate the environmental impacts that can result from agriculture, but they do not provide insight to many of the ecosystem services or disservices related to soil health or soil degradation, especially when only a specific product lifecycle is investigated (Boone *et al.*, 2019). Thus, any assessment of the environmental sustainability of different agricultural approaches should also consider the potential ecosystem services that can be provided by the farm as a whole, rather than solely assessing environmental impacts from food production.

Farms can provide a variety of ecosystem services to their environments and communities, such as protecting habitats for local wildlife and pollinators, improving biodiversity, contributing to carbon sequestration, aiding in erosion control and flood mitigation, provisioning food, fibre and fuel, and protecting heritage areas, and providing recreational, education, and employment opportunities (Swinton *et al.*, 2007; Power, 2010; Robertson *et al.*, 2014). However, farms can also negatively influence these functions, contributing to ecosystem *dis*services, if the land is not protected and managed appropriately (Lal, 2004; UK Environment Agency, 2019). Indeed, intensive agriculture is cited as one of the main contributors to soil degradation globally, as well as in the UK (Bai *et al.*, 2008; Graves *et al.*, 2011).

1.3.1 Soil health and related ecosystem services

Soils serve as an important natural capital asset from which ecosystem services flow to meet human needs (Dominati, Patterson and Mackay, 2010; Robinson *et al.*, 2013). Soils provide the basis for the provisioning of food, fibre, and some fuel sources; indeed, it is estimated that 95% of the global food supply is produced directly or indirectly on soil (FAO, 2015). In addition, soils are involved in regulating the climate, as well as water, atmospheric gas, and nutrient cycles; providing habitats for many micro-organisms and other wildlife; controlling erosion; and contributing to carbon sequestration, flood mitigation, water purification, pollutant and contaminant immobilisation, and the decomposition of organic waste (Daily, Matson and Vitousek, 1997; Dominati, Patterson and Mackay, 2010; Edmondson *et al.*, 2011; Natural Capital Committee, 2019). Soils are estimated to hold approximately three times as much carbon as the atmosphere (Lal, 2004); thus, they provide an important strategy toward climate change mitigation.

As soils provide the foundation for the majority of land-based ecosystem services (Dominati, Patterson and Mackay, 2010), protecting and regenerating agricultural soils is essential to ensure future food production and the continued provisioning of ecosystem services (Lal, 2015; IPBES, 2018; Wentworth and Tresise, 2022). The health of a soil is linked to its resilience, or its ability to recover from or adapt to stress (Lehman *et al.*, 2015). Healthy and resilient soils are thus a key part of a resilient food system and are essential for sustaining communities around the world, especially in the face of new environmental stresses associated with climate change.

Soil health has been defined as "the continued capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments, and maintain plant, animal and human health" (Doran and Safley, 1997), differentiating from 'soil quality' in the particular focus of soil as a living ecosystem (Karlen, 2012; Tully and McAskill, 2020). This definition of soil health clearly highlights the role that soils play in contributing to ecosystem services, although these concepts emerged in distinct fields. Soil health and quality frameworks emerged from soil science (Karlen *et al.*, 1997; Karlen, 2012), whilst the concept of ecosystem services emerged from ecology and economics as a way to place value on natural resources and the goods and services that they provide (Daily, 1997; Millennium Ecosystem Assessment, 2003). Recent research merges these concepts more eloquently, where soils are seen as the natural capital stock from which ecosystem services flow, and soil health frameworks provide indicators of the state of this natural capital stock at a given point in time (Robinson *et al.*, 2012). Viewing soil through an ecosystem service lens is important from a policy standpoint (Defra, 2011), because it provides a more obvious way to depict how improving soil health provides societal benefits (Robinson *et al.*, 2012).

1.3.2 Soil restoration priorities in UK policy

Although soil has been overlooked in UK environmental policy for many years (UK Environment Agency, 2019), the past decade has seen growing policy interest in soils due to interest in ecosystem services and concerns about soil degradation and declining carbon stocks (Wentworth and Tresise, 2022). Regenerating soils is seen as a pressing global priority, with UN bodies estimating that soil restoration processes must start within a decade in order to limit land degradation to a level that does not endanger the UN sustainable development goals (IPBES, 2018). In the UK, the Government has pledged to ensure soils are managed sustainably by 2030 (Defra, 2009; UK Government, 2018), with the recent Government's 25 Year Environment Plan highlighting that critical steps must be taken toward restoring UK soils (UK Government, 2018). In line with this, the UK Department for Environment, Food & Rural Affairs (Defra) is developing the Soil Health Action Plan for England with insight from farmers and other stakeholders; currently in its draft stags, this plan aims to provide a framework of actions to improve and protect soil health (Defra, 2021a) and has been highlighted as a crucial step toward Governmental aims to halt species decline by 2030 (Defra & Natural England, 2022).

As 71% of the UK's land area is used for farming (Defra, 2022b), it is clear that the agricultural sector will be one of the main groups driving soil restoration. Following the UK's withdrawal from the EU ('Brexit'), new agricultural policies are being implemented to replace the previously established EU Common Agricultural Policy, which provided basic payments to farmers depending on farm land area (Defra, 2020). New policies focus on using public money to pay farmers and land managers for public goods through the Environmental Land Management Schemes (ELMS), with emphasis on improving the environment, improving animal health and welfare, and reducing carbon emissions (Defra, 2020).

The Sustainable Farming Incentive (SFI) scheme is one part of ELMS and is currently in its pilot stages. In 2022, the programme's main focus was to encourage actions that improve soil health (Defra & Rural Payments Agency, 2022a). Action points required by arable and horticultural farmers in the scheme include completing a soil assessment, producing a soil management plan, testing for soil organic matter, adding organic matter to soil, and having green cover or a multispecies cover crop on the majority of land over the winter (Defra & Rural Payments Agency, 2022c). Future incentive areas are to be added in following years, which in 2023 may include areas such as nutrient management, integrated pest management, and hedgerow improvement, as well as potentially supporting agroforestry and organic farming in later years (Defra, 2022d).

Although there is contention from agricultural and sustainability groups about how to best implement ELMS, with particular concerns about sufficient financial support for farmers and the rigour of environmental standards (NFU, 2021b, 2021c; Sustain, 2021; AHDB, 2022a; Huang *et al.*, 2022), it is clear that the UK Government is recognising the crucial role that soil health plays in food production and the provisioning of other ecosystem services. As the SFI is requiring farmers and land managers to test for soil quality, it is important that research informs the metrics that are chosen to relate to the provisioning of public goods. Additionally, as the SFI is providing payments for specific practices, such as cover cropping and potentially organic agriculture in the future, it is more critical than ever to investigate different models of sustainable farming (e.g., organic or conventional) and land management (e.g., land sparing vs. sharing) to identify trade-offs with certain practices that may contribute to both ecosystem services and disservices.

1.3.3 Soil health, degradation, and metrics

Although agriculture can contribute to many ecosystem services, it also relies upon many of the services supported by soil processes for crop production, such as nutrient cycling, water filtration, erosion control, and maintaining soil structure, among others (Swinton *et al.*, 2007; Zhang *et al.*, 2007). As previously highlighted, the mismanagement of soils can lead to soil degradation, which is one of the main challenges facing future food production (Gomiero, 2016; Jeffery and Verheijen, 2020). This issue was highlighted by the UK's former environmental and agriculture Secretary of State, Michael Gove, who warned that there may only be 30-60 harvests left for farms in some parts of the country due to the loss of soil fertility (Sustain, 2017; van der Zee, 2017).

In England and Wales, soil degradation has been driven mainly by the loss of organic matter, compaction, and erosion, and to a lesser extent by soil biota loss, contamination, and soil sealing (Graves *et al.*, 2011). In 2010, it was estimated that these processes cost £1.2 billion GBP in England and Wales each year, stemming mainly from the loss of ecosystem service provisioning supported by soils, especially relating to agricultural production, flood mitigation, water purification, and carbon sequestration (Graves *et al.*, 2011).

Soil health metrics provide indicators of the state or quality of the soil and thus relate to its ability to contribute to ecosystem services; many soil health metrics are therefore used interchangeably as ecosystem service indicators (Lehmann *et al.*, 2020). Soil health metrics are comprised of a variety of physical, chemical, and biological soil properties, with much research currently dedicated to understanding the relative importance of certain properties and creating a simplified and standardised set of indicators that are easy to measure and are sensitive to changes in land management (Schindelbeck *et al.*, 2008; Cardoso *et al.*, 2013; Lehmann *et al.*, 2020). Despite a wealth of research identifying different soil health metric frameworks (Stewart *et al.*, 2018), there is still no commonly agreed upon set of metrics among the scientific community, making policy incentivising soil health practices more difficult (Jeffery and Verheijen, 2020). This has been a key challenge of ELMS policy in the UK, with the UK Government designating at least £200,000 to develop and test appropriate soil health metrics (UK Government, 2018).

Although there is contention in the scientific community about the best soil health parameters to test for particular purposes, basic soil health/quality indicators are continuously reported among studies. In 2017, the Soil Health Institute endorsed 19 Tier 1 soil health indicators, which include measures such as bulk density, pH, soil organic matter (SOM) or soil organic carbon (SOC), nitrogen (N), phosphorus (P), potassium (K), water-holding capacity, micronutrients, texture, and aggregate stability, among others (Soil Health Institute, 2017). The prior five were also identified as the most frequently cited soil quality indicators among UK soil scientists by the National Capital Committee, a former advisory committee to the UK Government (Natural Capital Committee, 2019). Hazardous heavy metal concentrations (such as Pb, Cd, As, and Hg) have also been highlighted as important measures in potentially contaminated soils, such as urban soils (Schindelbeck *et al.*, 2008; Cardoso *et al.*, 2013). These soil health indicators can then be used as proxies to indicate the ecosystem services that relate to them; for example, nutrient content and soil fertility, or water-holding capacity and flood mitigation (Robertson *et al.*, 2014; Eastburn *et al.*, 2017). The following sections further explore various soil health metrics in relation to ecosystem services and pressing UK soil degradation challenges.

1.3.3.1 Carbon sequestration and organic matter

With the UK aiming to reach net zero emissions by 2050, maintaining soil organic carbon stores is more critical than ever (BEIS, 2021c). Globally, soils store more carbon than is contained in both the atmosphere and terrestrial vegetation combined (FAO and ITPS, 2015). In the UK, soils currently store about 10 billion tonnes of carbon, roughly equal to 80 years of current annual UK greenhouse gas emissions (Defra, 2009; UK Environment Agency, 2019).

Soil organic carbon (SOC) is often seen as the defining constituent of soil, being the single most important and reliable indicator for monitoring soil health or degradation (Manlay, Feller and Swift, 2007; Rajan *et al.*, 2010). SOC represents a major constituent (55-60% by mass) of soil organic matter (SOM), which refers to any material produced by living organisms and undergoing decomposition in the soil (Bot and Benites, 2005). Thus, the use of SOM and SOC as soil health indicators is seen to be essentially synonymous.

SOC makes an obvious contribution to carbon sequestration; however, much of SOC is stored in labile carbon pools that will be decomposed and released back into the atmosphere over short time-frames, from a few days to a few years (Clara *et al.*, 2017). Thus, measures of SOC do not fully equate with long-term carbon storage, with current research investigating how to retain carbon in slower SOC pools by potentially increasing the SOC stocks in subsoils (Lorenz and Lal, 2005; Rumpel, Chabbi and Marschner, 2012).

SOM also contributes to many other soil processes and ecosystem services besides carbon sequestration. SOM is involved in the binding of soil particles and is thus essential for aggregate formation and defining soil structure (Bot and Benites, 2005). It influences aggregate stability, water-holding capacity, and water infiltration rates, and therefore is important for ecosystem services such as minimising surface runoff, increasing a soil’s resiliency to drought, and resisting erosion (Watts and Dexter, 1997; Bot and Benites, 2005; Robinson *et al.*, 2013; Clara *et al.*, 2017). Additionally, SOM holds a store of essential nutrients for plant growth as it is composed mainly of decomposed plant tissue (Bot and Benites, 2005); thus, it has been positively linked to soil biodiversity by serving as a food source (Bot and Benites, 2005; Clara *et al.*, 2017). Because SOM and SOC support soil structure and biological activity, increases in SOC have also been positively correlated with increases in crop yields (Bennett *et al.*, 2010; Lal, 2010).

Although it is clear that many ecosystem services flow from SOC, this resource is under threat in the UK due to losses of organic matter and erosion. Land use change and agricultural activities such as tillage and drainage can enhance the mineralisation of soil organic carbon and release CO_2 into the atmosphere (Lal, 2004). Globally, it is estimated that land use transformations to cropland have resulted in SOC losses of 40-60% (Guo and Gifford, 2002). Erosion is another important soil degradation process, which has the most severe impact on depleting the SOC pool (Lal, 2004). When erosion exceeds the rate of soil formation, soil is lost (UK Environment Agency, 2019). The UK Environment Agency highlighted that intensive agriculture has contributed to an increase in erosion, related to tillage practices and overgrazing, as well as the increase in field size and consequent decreased length of hedgerows that can protect fields from wind and water (UK Environment Agency, 2019). Every year, England and Wales loses 2.9 million tonnes of topsoil to erosion; it is estimated that 17% of arable soils have already been eroded, with an additional 13% (2 million ha) being at risk (Graves *et al.*, 2011; UK Environment Agency, 2019).

Additions of organic matter to soil are thus crucial for maintaining soil structure and SOC stocks (Magdoff and Weil, 2004). Research shows that biodynamic or organic farms generally have higher SOM levels than conventional farms, due to higher and regular additions of organic matter (e.g., through composts, manure, and cover crops) (Stockdale *et al.*, 2001; Shepherd, Harrison and Webb, 2002). Rates and depths of tillage can also affect SOM/SOC concentrations in the topsoil (Shepherd, Harrison and Webb, 2002; Magdoff and Weil, 2004), with research showing that minimum or no tillage systems generally have higher levels of SOM and SOC (particularly in the topsoil layer) compared to conventional or intensively tilled systems (Haddaway *et al.*, 2017).

1.3.3.2 Compaction and flood mitigation

Flood risk has been identified as one of the UK's top climate change risk areas due to the potential increasing severity of extreme rainfall events (UK Government, 2016; Committee on Climate Change, 2017). On agricultural landscapes, high rainfall events are an issue because they can lead to soil erosion and nutrient losses that pollute nearby areas and waterways through runoff (Antolini *et al.*, 2020). This has led to heightened focus on ways to enhance flood mitigation through the utilisation of natural landscapes, known as natural flood management (UK Government, 2018; Wentworth and Zu Ermgassen, 2020).

Increasing soil permeability and water-holding capacity is an important part of natural flood management, so that water is infiltrated more readily into the soil rather than contributing to runoff (Yang and Zhang, 2011; Velickov *et al.*, 2014). Thus, water-holding capacity and soil infiltration rates are seen as important measures indicating the capacity of soil to contribute to flood mitigation, and consequently reduced runoff and erosion (Lovell and Taylor, 2013; Keesstra *et al.*, 2018).

Flood mitigation is also related to soil porosity (Vári *et al.*, 2022), where higher pore space is usually associated with increased water permeability and infiltration (Helalia, 1993). Measures of bulk density, or the ratio of oven-dried soil mass to bulk volume, can be used to indicate total pore space; lower bulk densities generally equate to higher pore space (Dominati, Patterson and Mackay, 2010; Edmondson *et al.*, 2014; Clara *et al.*, 2017). Thus, bulk density is often negatively correlated to water-holding capacity (Libohova *et al.*, 2018).

Bulk density is also used as an indicator of compaction in soils, where higher values are associated with more compacted soils (Edmondson *et al.*, 2011). Compaction is the physical reduction in soil volume due to compressive forces, which results in reduced soil pore space (Graves *et al.*, 2011). Compaction reduces the capacity of soils to provide essential ecosystem services, such as crop production and flood mitigation. Compaction can impair root growth and also limits the ability of soils to hold and filter water, thus increasing the risk of surface runoff and flooding (UK Environment Agency, 2019). Soil waterlogging that may occur as a result of compaction can also lead to the release of greenhouse gases from the soil, particularly nitrous oxide and methane (Graves *et al.*, 2011; Wang *et al.*, 2017).

Heavy machinery, wheel traffic, and intensive grazing are some of the main causes of soil compaction on farms (Shah *et al.*, 2017; Dejong-Hughes and Daigh, 2022). In 2010, 25% of soils in England and Wales were estimated to be at risk of compaction, or approximately 3.9 million ha (Graves *et al.*, 2011). Soil compaction is especially an issue in urban soils, which have often been disturbed or displaced, have relatively low organic matter, and may be subject to industrial activities or surface traffic, all of which destroy soil structure (Pouyat *et al.*, 2010; Beniston and Lal, 2012).

SOM is often correlated with higher water-holding capacity and lower bulk density (Libohova *et al.*, 2018); thus, increasing organic matter addition to soil is one way to improve soil structure and permeability on farms, reducing runoff and erosion (Magdoff and Weil, 2004; Kremen and Miles, 2012). For urban soils in particular, additions of compost have been an effective way to improve physical soil properties and reduce compaction (Beniston, Lal and Mercer, 2016; Kranz *et al.*, 2020). Reducing tillage, employing cover crops, using appropriate equipment, and controlling field traffic have been cited as additional strategies to limit compaction and improve soil water infiltration (Blanco-Canqui *et al.*, 2015; Antolini *et al.*, 2020; Dejong-Hughes and Daigh, 2022).

1.3.3.3 Soil nutrients and fertility

Crop nutrition is an essential part of soil fertility (Cooke, 1967), which specifically relates to the ability of the soil to support crop production (Patzel, Sticher and Karlen, 2000). Growing crops consistently depletes the nutrient status of soils, which is why nutrient inputs must be continuously added to agricultural soils (Goulding, Jarvis and Whitmore, 2008). Effective nutrient management results when soil nutrient levels match crop nutrient needs (Johnston and Bruulsema, 2014). Excess nutrients can be leached from soils, leading to nutrient loading and habitat destruction in other ecosystems (Lal, 2015; UK Environment Agency, 2019). On the other hand, a deficiency in just one nutrient can limit crop growth (Black, 1993; Marschner, 2012), and consistent under-fertilisation over time can lead to ‘nutrient mining’ and soil degradation (Lal, 2004). SOM is an important part of soil fertility because it acts as a reservoir for many essential nutrients, such as nitrogen (N), which is typically the limiting nutrient for crop growth (Goulding, Jarvis and Whitmore, 2008). SOM also retains ionic forms of nutrients that can be rapidly exchanged into the soil solution for crop uptake (Berry *et al.*, 2002). Microorganisms are essential to the release of nutrients in plant available forms from SOM reserves (Beare *et al.*, 1995), thus highlighting the importance of soil biological factors in contributing to soil fertility.

Macronutrient (N, P, K, S, Ca, Mg) and micronutrient levels (Fe, Mn, B, Cu, Zn, Mo, and Cl) are commonly assessed within soils to understand their relation to crop growth (Schindelbeck *et al.*, 2008). However, it should be noted that total nutrient content does not equate to the nutrients availabile to crops, as most nutrients are held in insoluble organic and mineral forms (Jarvis *et al.*, 1996; Stockdale *et al.*, 2002). Thus, total nutrient content will also not necessarily relate to the propensity of nutrients to be leached from a system. Still, evaluating total nutrient content provides insight to the soil nutrient reserves and any potential excesses or deficiencies.

The measure of total N and SOC can also be used to derive carbon-to-nitrogen ratios (C/N ratio), which are useful to assess the balance of two soil elements that are related to nutrient availability and microorganism activity. Low C/N ratios are generally associated rapid N mineralisation by microbes; more N is available for plant use, which generally results in high plant growth, but can also potentially lead to nitrate leaching (Dungait *et al.*, 2012). On the other hand, high C/N ratios are associated with low N levels; when these levels are too low to support increased microbe growth, microbes will consume N from the soil and immobilise it in their tissues, leading to potential N deficiencies for crops (USDA NRSC, 2011; Dungait *et al.*, 2012).

Nutrient management differs in organic and conventionally-managed farming systems because of the different types of fertiliser inputs (Watson, Atkinson, *et al.*, 2002). Even if total soil nutrient concentrations do not differ between these systems, the proportion of nutrients in various soil pools and the degree of nutrient availability often does (Stockdale *et al.*, 2002). In conventional systems, nutrient deficiencies are mainly overcome through the application of mineral fertilisers, which tend to have relatively high proportions of soluble minerals that are readily available to crops (Watson, Atkinson, *et al.*, 2002). In organic farming systems, however, the use of mineral fertilisers is restricted. Only organic nutrient amendments may be added to the soil, such as manures, composts, and cover crops / green manures, where nutrients are mainly in insoluble or organic forms (Stockdale *et al.*, 2002; Watson, Atkinson, *et al.*, 2002). Thus, soil fertility in these systems relies on a long-term approach that often integrates the rotation of crops with varying nutrient requirements (Watson, Atkinson, *et al.*, 2002), drawing upon the slow release of nutrient reserves, such as from SOM mineralisation.

However, the generally low N contents and slow mineralisation rates of organic fertilisers make N availability a concern for organically-managed soils because of possible limiting impacts on crop production and yields (Berry *et al.*, 2002). Another concern for organic farming is the possible 'mining' of soil phosphorus (P) and potassium (K) reserves that may have been built up from previous conventional production; if there are insufficient additions of available P and K from organic inputs, this could impact future soil fertility. Indeed, long-term trials comparing organic and conventional arable farming in the UK have found that organically-managed land tends to have lower levels of extractable P and K (Gosling and Shepherd, 2005).

In terms of N losses through leaching and runoff, differences between organic and conventional systems are generally low and likely depend more on the timing of cultivation and fertiliser application than the type of farming system (Stockdale *et al.*, 2002). Although P leaching is usually seen to be less of a concern than N leaching, this can occur in soils with high P concentrations (Kleinman *et al.*, 2011). High P concentrations can result from long-term over-application of P to soils, often seen with excessive manure application (Stockdale *et al.*, 2002; Kleinman *et al.*, 2011; Powlson *et al.*, 2011). P losses can also occur through surface run-off, which is not related to P application amounts (Stockdale *et al.*, 2002); thus, erosion control on farms, such as maintaining soil cover, is also essential to limit leaching.

P nutrient losses have been cited as a particular area of concern for urban agriculture (Small *et al.*, 2019). Urban farms and gardens are often characterised by high nutrient application rates (especially from compost) and low nutrient use efficiencies; many studies have found that N and especially P application rates in urban agriculture exceed crop demand (Abdulkadir *et al.*, 2013; Small *et al.*, 2019; Wielemaker *et al.*, 2019). The accumulation of P may be a result of applying organic inputs, such as compost and manure, to satisfy crop N requirements; however, since these inputs often have a low N:P ratios, and also low amounts of readily available N, this can lead to excessive total N and P levels in the soil (Kleinman *et al.*, 2011; Small *et al.*, 2019). The contribution that excess total N and P concentrations make to leaching is not clear, as this also relates also to nutrient solubility and potential erosion; however, this presents an important area of future research regarding the sustainability of urban farming systems.

1.3.3.4 Contamination and heavy metals

Although contamination of UK soils has been mainly seen asa past issue, the UK Environment Agency has recently highlighted new concerns (UK Environment Agency, 2019). Soil contaminants are an area of worry because they can be toxic at certain levels and therefore can be harmful to the health of plants, humans, and other wildlife. Adverse impacts can occur from direct contact with soil, as well as from exposure to contaminants transferred through waterbodies and into the food chain (Chaney, Sterret and Mielke, 1984; UK Environment Agency, 2019). Regarding agricultural land, heavy metals and other contaminants can be introduced to the soil via atmospheric deposition, pesticide application, and the spreading of manures, treated sewage sludge, and other wastes (Nicholson *et al.*, 2003). Compost has been cited as both a potential source of heavy metals to agricultural soils (UK Environment Agency, 2019), but also as a solution for immobilising them (Brown and Jameton, 2000; Kumar and Hundal, 2016). Additional pollutant sources include emissions from various industrial, transport, energy, and waste sectors; however, these are usually seen to be more of a concern for urban soils, due to larger discharge intensities (Li *et al.*, 2018). Thus, testing for heavy metal contamination is often recommended for urban soils prior to growing food (Schindelbeck *et al.*, 2008; Cardoso *et al.*, 2013).

Heavy metals in soils can be transferred to humans through direct ingestion, skin contact, inhalation, or ingestion of a contaminated crop (Oliver and Gregory, 2015). Heavy metals can contaminate the surface of produce through direct contact, dust, suspension, or rain-splash (Lehmann *et al.*, 2020); some metals or metalloids, like arsenic (As), can also be taken up through plant roots and accumulate in the edible portion of the crop (Oliver and Gregory, 2015). Certain metals are actually essential to both plants and animals in trace amounts, such as zinc (Zn), copper (Cu), nickel (Ni), and cobalt (Co); these serve as essential micronutrients, and, in mammals, are often involved in enzyme functioning (Oliver and

Gregory, 2015). Arsenic, typically seen as a particularly ‘poisonous’ element, may also have a role in metabolic functioning in very trace amounts (Hunter, 2008). Iron (Fe) is also essential to humans, and high levels in the soil are not necessarily seen to be a particular concern for human health (Oliver and Gregory, 2015; Okereafor *et al.*, 2020); however, high levels can be concerning for crop production because this can lead to the development of a hard soil layer (iron pan) that can prevent root penetration, usually in cases of declining soil fertility (Cunningham, Collins and Cummins, 2001). Finally, there are also heavy metals that are not necessary for animal or plant life, including mercury (Hg), cadmium (Cd), and lead (Pb) (Okereafor *et al.*, 2020). At high concentrations, all of these elements can pollute the environment and result in negative health impacts for humans, plants, and animals.

Although high levels of heavy metals may be present in urban soils and pose potential health risks, it does not necessarily imply that crops grown in these soils will be adversely affected or that the crops will have toxic levels of accumulated heavy metals (Gibson and Farmer, 1986; Crispo *et al.*, 2021; Ziss *et al.*, 2021). Heavy metals can be immobilised by organic matter, such as compost, (Lepp, 1981), and generally the bioavailable concentration that can be taken up by plants is relatively low (Crispo *et al.*, 2021). However, even if contaminants are not taken up by the crop, they can still exist on the crop surface due to deposition; a study in Bologna, Italy found that heavy metal concentrations on crop surfaces were highest for those grown near road-ways, whilst levels of heavy metals in plant tissue were similar between urban and rurally-grown crops (Antisari *et al.*, 2015). As exposure can also occur through other routes besides ingestion of contaminated crops, it is thus important to assess potentially toxic metal concentrations in the soils of urban farms and gardens to minimise any health risks (Chaney, Sterret and Mielke, 1984).

1.4 Experiences of food insecurity and community responses during COVID-19: A case study in the UK

Although important, analyses of agricultural production (as described in Sections 1.2 and 1.1) do not provide a full picture of the sustainability and resiliency of the entire food system. Following the prior definition provided in this thesis (Section 1.1.2), a sustainable food system must be one built on social justice that enables food security to be ubiquitously achieved (Sumner, 2011). In the face of crisis, a resilient food system must be able to adapt to shocks and stresses and recover quickly in order to maintain food security (GFS, 2019). Although farmers and food producers might provide an adequate and consistent *supply* of food, many are still left hungry or malnourished, even in wealthy countries (Steiner, Geissler and Schernhammer, 2019). Thus, following a food systems approach, it becomes necessary to also examine the other side of the food supply chain – the consumer.

In this thesis, the sustainability and resiliency of food systems in relation to issues of food security and access are explored during a key time of crisis – the COVID-19 pandemic. This pandemic critically impacted food systems around the world, which sparked much conversation and debate on how to increase resiliency of these systems to ensure future food security (Aday and Aday, 2020; Béné, 2020; Garnett, Doherty and Heron, 2020; Bellamy *et al.*, 2021). The COVID-19 pandemic created wide-ranging barriers to achieving all four dimensions of food security (as defined in Section 1.1.1.1): food availability, accessibility, utilisation, and the stability of the prior three dimensions over time. The ways that COVID-19 diminished the resources available to people to achieve food security and cope with crisis are highlighted, whilst also acknowledging how these resources were already disappearing before the crisis.

1.4.1 Resources required for food security

Discussing the first three dimensions of food security in more detail, physical availability can be understood as the 'supply side', related to whether or not there is a sufficient amount of affordable, nutritious, and culturally appropriate food available in the places where people live (FAO, 2008). Accessibility of food refers to whether people can get to this food (e.g., physical ability to walk to the shop and carry food home or the ability to afford transport) and afford to buy it. These first two dimensions are the primary focus of research into household food insecurity in wealthy nations, seen within the predominant dialogue on 'food deserts' as areas of low food access (Baker *et al.*, 2006; Jetter and Cassady, 2006; Shaw, 2006; Hilmers, Hilmers and Dave, 2012) and 'food poverty' in relation to the ability to buy food (Dowler, 2002; Dowler and O'Connor, 2012; Riches and Silvasti, 2014; Williams *et al.*, 2016; Long *et al.*, 2020).

The third dimension, utilisation, is typically considered through the angle of food safety and nutritional health (Barrett, 2010; Battersby, 2011); in wealthy countries, this is generally addressed through educational interventions about good dietary practice (Koutoukidis and Jebb, 2019). Following these framings, we see that food security should be possible if people have appropriate and affordable food shops in their area that stock safe and nutritious food; have enough money to afford food, the transport to get it, and the facilities to cook it; and have the 'know-how' to ensure that they are getting a well-balanced and nutritious diet. However, we see that this conceptualisation misses out on critical components that have been known to be bidirectionally associated with food insecurity – namely, health. Thus, this research proposes that the 'food utilisation' dimension is conceptualised more broadly to include the physical and mental capabilities needed to shop for food, bring it home, cook it, and eat it. When physical and mental health are viewed as co-determinants of food insecurity, rather than just merely effects, more holistic pathways that address the entire complexity of food insecurity can be promoted. It thus becomes clear that the resources needed to achieve food security are not solely related to food supply and financial resources, but also include the resources that promote and protect physical and mental health.

1.4.2 Resilience to crisis through social and place-based resources

Times of personal crisis create barriers to meeting all four dimensions of food security. As the resources needed to achieve one dimension (e.g., income) are stretched, increased strain is placed on other resources (e.g., accessing healthy foods), demonstrating how these dimensions overlap and influence each other (Burns *et al.*, 2011; Begley *et al.*, 2019). The resources necessary to achieve food security include those on an individual level (income and mental and physical capability), but also those related to place (e.g., local services, available shops, and community groups) and relationships (social networks and social trust) (Mathie and Cunningham, 2003; Blake, 2019). The presence of these resources and ability to identify and mobilise them builds resiliency as they provide the opportunities to adapt, overcome, and recover from crisis or avoid it entirely (Aranda and Hart, 2014; Blake, 2019). This research shows how COVID-19 both erased these resources and the ability to draw upon them, but in this section further context is provided for the argument that these resources were already being narrowed before COVID-19 through neoliberalist ideologies, ableism, and austerity policies, which further limited resiliency to cope with the crisis.

This section thus provides context on the social resources that build resilience to crisis, explored through the idea of 'social capital.' Further context is provided on the ideas of neoliberalism, austerity, and ableism. This background is provided to support a main argument of this research: that austerity policies (disinvestment in public services), internalised neoliberal ideologies ('the responsible individual'), and ableism have narrowed the resources available to people to cope with food insecurity and crisis and also made them more difficult to access. The research in this thesis explores this concept through the lived experiences of food insecurity through the COVID-19 crisis.

1.4.2.1 Social capital: expanding resources

In addition to the individual and place-based resources needed to achieve food security are relational ones. The view of social networks as a critical human resource is not new, and in some cases has actually been attributed to our success as a species. Indeed, it has been viewed that the inherent nature of humans to engage in large social groups and act collectively is what allowed the human species to rise and prosper in this world (Dunbar, 1998; Helliwell, Huang and Wang, 2014). This builds on the idea that humans are not just social creatures but are "pro-social," getting happiness from being with others but also from doing things both with and for others (Batson and Shaw, 1991). This underlies an inherent want to be together and help each other.

Filtering this evolutionary approach down to today, the view still holds that social connections and social support are necessary resources to living a happy and healthy life.

This idea has been purported widely through the idea of "social capital", defined here as the human resources that reside in social networks (Li, 2007). Although various theories exist on what social capital entails and how it should be measured, it can broadly be thought to include both informal social networks (e.g., personal ties with relatives, friends, neighbours, and acquaintances) and formal social networks (e.g., membership in social groups, voluntary organisations, or other non-governmental organisations) (Granovetter, 1973; Bourdieu, 1993; Putnam, 1995, 2000; Li, Pickles and Savage, 2005; Li, 2007). The informal networks formed through personal ties provide emotional and practical resources in the ways that people share feelings, give each other attention, cultivate understanding and trust, offer advice and help with problems, share responsibilities, and provide social, material, and financial support (Lin, 2001; Aldrich and Meyer, 2015; Bian, Hao and Li, 2018). Formal networks are sources of institutionalised resources, such as information and status, as well as skills and assets (Y. Li, 2015). Social capital is thus linked to place through the availability of public, social, and community spaces and existing groups and organisations that allow for the formation of social ties (Klinenberg, 2003; Cattell *et al.*, 2008).

Social networks can both provide critical resources in times of crisis and also expand the types of resources available to an individual, outside of a particular place or circumstance; in this way, social capital has been shown to underly both individual and communal resilience (Mathie and Cunningham, 2003; Aldrich and Meyer, 2015). A wide array of research has shown how social capital contributes to positive health, wellbeing, and overall life satisfaction (Greenblatt, Becerra and Serafetinides, 1982; Putnam, 2000; Li, 2007; Helliwell, 2011), which builds individual capabilities to cope with crises like food insecurity, (e.g., through better physical and mental health).

Social capital and social trust is also important for building community and thus for community resilience in times of crisis (Mathie and Cunningham, 2003; Helliwell, Huang and Wang, 2014; Aldrich and Meyer, 2015). There is evidence that communities and countries with better social capital and trust respond to crisis more happily, quickly, and effectively and that this is not always necessarily dependent on socioeconomic conditions (Nakagawa and Shaw, 2004; Helliwell, Huang and Wang, 2014; Aldrich and Meyer, 2015). Helliwell, Huang and Wang (2014) provide the example of South Korea during the 2007-8 financial crisis; the country exhibited a fast economic recovery and also some of the highest wellbeing levels globally post-crisis. The South Korean president at the time, Lee Myung-bak, attributed this to the way in which people and companies "decided to share the burden", as employees accepted salary cuts and companies accepted reduced profits so that less people lost their jobs (Helliwell, 2011). This cooperative approach that benefitted the 'greater good' was both facilitated by, and contributed to, higher social capital and engagement. Drawing on the work of Ostrom (1990), Helliwell, Huang and Wang (2014) further argue that the quality of social capital is linked to a society's ability to discover and implement better solutions to resource conflicts in ways that can facilitate cooperative strategies. Social connections built on social trust and reciprocal relationships thus become a vital platform to pool and share resources to adapt, overcome, and transform to crisis.

1.4.2.2 Neoliberal-ableism and austerity: narrowing resources

As the scholarship around social capital builds the idea of resilience through relationships, neoliberalism and ableism narrow resilience to an individual responsibility and asset (Joseph, 2013). Although interpreted in many ways by a variety of disciplines, 'neoliberalism' here refers to policies and ideologies that focus on individual entrepreneurship and freedoms and promote market deregulation and privatisation (Thorsen and Lie, 2006; Venugopal, 2015). Through this lens, the purpose of the state is to safeguard individual, commercial, liberty, and private property rights, thus corresponding to a withdrawal of the welfare state (Thorsen and Lie, 2006; Venugopal, 2015).

Neoliberal policies and ideologies have trickled down to a personal level, creating the view of a 'neoliberal subject' as free, autonomous, and personally responsible for their own social and economic wellbeing (Brown and Baker, 2013; Joseph, 2013; Wrenn and Waller, 2017). In times of crisis, the responsibility is placed onto the individual to make appropriate decisions and find ways to "bounce back"; thus, neoliberal crisis responses are focused on individual preparedness, informing people to make better decisions, and highlighting individual roles and responsibility (Joseph, 2013). The notion of individualised responsibility tends toward ideas of competition within social life (Foucault, 2008), as opposed to shared or communal responsibilities.

Ableism, often studied in the dis/ability context, refers to a set of beliefs and practices that project the "species-typical" citizen, or norm, as one who is able-bodied, ready to work, productive, and contributing (Goodley, 2014; Goodley and Lawthom, 2019). This bounded norm thus constructs a space such that people outside of it, i.e. people with disabilities, are viewed as 'others' (Chouinard, 1997) or as a "diminished state of being human" (Campbell, 2001). Ableism is therefore essentially a framework for stigma, where stigma is broadly defined as social devaluation based on perceiving someone, or oneself, as deviant to a social norm (Bogart and Dunn, 2019). Ableism defines what that norm should be.

As neoliberalism merges with ableism to 'neoliberal-ableism', this provides the context for how these ideas become concrete barriers to accessing various resources available to cope with crisis. Neoliberal-ableism brings forth the idea of the human 'norm' or 'desirable person' as completely autonomous, healthy, rational, and reasonable – and therefore un-needing, self-contained, and isolated (Goodley and Lawthom, 2019). As this desirable personhood is normalised and internalised by people (Goodley, Lawthom and Runswick-Cole, 2014), any deviation from this norm can induce stigmatisation by others but also self-stigmatisation, which involves the embodiment of a spoiled social identity by perceiving oneself as socially different (Goffman, 1963; Chan, Stoové and Reidpath, 2008; Bos *et al.*, 2013). As these feelings of stigma and aloneness set in (Dolan, 2021; Thomas, 2021), this can prevent people from seeking or accepting help and drawing upon social capital (van der Horst, Pascucci and Bol, 2014; Caplan, 2016) whilst also negatively impacting health, thus narrowing the resources available to people to cope with crises such as food insecurity and undermining their ability to use them.

As neoliberal ideologies have been channelled internally to limit the resources available to people within their personal networks, austerity measures have further narrowed the resources available to people within the wider state structure. Austerity policies can be viewed as an extension of neoliberal ideology, characterised by a disinvestment in the public sector which seeks to reduce the 'welfare state' in favour of market development (Arrieta, 2022). Austerity policies are supported by some who believe disinvestment in the public sector allows these organisations to become more efficient in their service provision and that increased privatisation allows for more public choice; on the other hand, some refute these policies by highlighting how they limit the ability to provide basic services to those most vulnerable through the reduction of the public sector and rise of profit-seeking companies (Taylor-Gooby and Stoker, 2011; Arrieta, 2022).

Austerity policies in the UK have long been highlighted as contributing to rising food insecurity and limiting the resources available to people to cope with it (Jenkins *et al.*, 2021). These policies came to the forefront in 2010, as the Conservative-Liberal Coalition Government aimed to reduce the economic deficit as quickly as possible through reduced public spending and massive welfare reform (Dowler and Lambie-Mumford, 2015; Briggs and Foord, 2017). Practically, this was seen through changes to tax, social security, and welfare benefits as well as through reductions in funding for local support services, such as child care and day centres, and central government services (Wood, 2012; Dowler and Lambie-Mumford, 2015). These reforms highlighted a restructuring that shifted responsibilities from the state to the private sector and individual citizens (Williams, Goodwin and Cloke, 2014; Briggs and Foord, 2017).

With these cuts came the rise of the voluntary sector to fill the gap, perhaps seen most starkly in the dramatic rise of food banks which became the default welfare safety net in the UK post-2010 (Caplan, 2016; Sosenko, Bramley and Bhattacharjee, 2022). The number of food banks within the Trussell Trust network, estimated to comprise approximately 55-60% of all UK food banks (Loopstra *et al.*, 2019; Sosenko, Bramley and Bhattacharjee, 2022), rose from just 35 in 2010/11, to 650 in 2013/14 and over 1,400 in 2020/21 (Sosenko *et al.*, 2019; Bramley *et al.*, 2021; The Trussell Trust, 2022b). Throughout all these years, the Trussell Trust has cited again and again how welfare and benefit cuts have pushed people into severe food insecurity, whilst also highlighting the difficulty in accessing already-stretched formal support measures (e.g., mental health services) (Bramley *et al.*, 2021). Research has shown how austerity measures such as reduced value of welfare benefits, the structure of existing benefits, taxes in social housing, and benefit sanctions have specifically driven foodbank use in the UK (Sosenko, Bramley and Bhattacharjee, 2022). This shows how austerity measures have diminished the resources available to people in the UK to cope with food insecurity and coinciding health and life challenges.

These difficulties in accessing an already barren landscape of state support, due to nearly a decade of austerity measures, was brutally brought to light with the COVID-19 pandemic. In a complete reversal of tone, the British government responded to the crisis by releasing unprecedented levels of expenditure to support health services, businesses, and communities. However, Arrieta (2022) argues that this could not undo the previous years of austerity, which had reduced the capacity of public institutions and households to cope with a shock such as COVID. This was seen in hospital bed shortages and lack of personal protective equipment (PPE) within the National Health Service and adult social care facilities, which was associated with increased deaths of staff in the early days of the pandemic (Daly, 2020; Dyer, 2020). In cases of benefits support, it was seen that the most vulnerable were still left out of increased support or were unable to get the support they needed, particularly low-income households, disabled people, and carers (Caplan, 2020; Arrieta, 2022).

This context thus provides the background for a key argument of this research – that austerity and neoliberal-ableism have narrowed the resources available to people to achieve food security and thus diminished resiliency in times of crisis. Austerity policies have narrowed financial and place-based resources, whilst stigma incited by neoliberal-ableism serves to further diminish the ability to access or utilise the resources that do exist or to draw upon existing social capital. This is further contextualised through the rise of food insecurity during the COVID-19 pandemic.

1.4.3 Food insecurity during COVID-19

In early March 2020, the World Health Organization (WHO) officially declared the rapid spreading of the COVID-19 virus as a pandemic (Cucinotta and Vanelli, 2020). To contain the spread of the virus, many national governments imposed lockdown measures. In England, the first lockdown began on 23 March 2020, with most restrictions lifted in early July; this lockdown is the focus of this study. Two more lockdowns were also later implemented in November 2020 and then again in January-March 2021 (Brown and Kirk-Wade, 2021). These lockdowns involved schools shutting, non-essential shops closing, and people being permitted to leave their home only for exercise, work, medical appointments and to acquire basic essentials (e.g., buying food). During the first COVID lockdown, the Government initially advised 1.5 million people in England to 'shield' (avoid leaving the home for any reason) for 12 weeks, due to being at extreme risk of hospitalization from the virus ('clinically extremely vulnerable') (UK Government, 2020a). This guidance was later updated to apply to a total of 2.2 million people in England, who were advised to shield until the end of July (UK Government, 2020b).

The shock of the pandemic and resulting lockdown restrictions soon devastated the food system. Although the global food system had experienced a variety of shocks in the past (e.g., financial crisis, droughts, etc.), the crisis of COVID-19 was unique in how it affected multiple sectors within the food system all at once, thus bringing to light a wide range of vulnerabilities all at the same time (Hendrickson, 2020). This included supply issues, as many farms, food processors, and distribution companies were unable to access necessary resources, such as labour; logistical problems throughout the international supply chain, which became deadlocked and unable to cope due to a reliance on a concentrated set of suppliers; and of course, financial and physical access issues for consumers, which coincided with a loss of income, self-isolation requirements, and a lack of available food (Garnett, Doherty and Heron, 2020; Petetin, 2020; Power *et al.*, 2020).

These combined challenges resulted in a rise in UK household food insecurity, creating a group of newly food insecure people and deepening the hardship for those already food insecure (Loopstra 2020; The Trussell Trust 2020). The Food Foundation estimated that food insecurity doubled with the onset of the pandemic, from nearly 8% pre-COVID to nearly 16% during the first two weeks of lockdown (The Food Foundation, 2021); other preliminary research suggested that rates quadrupled over the March-April period (Loopstra, 2020). The main drivers of UK food insecurity during this time were related to the inability to afford food; the need to self-isolate, and thus inability to physically access food; and the lack of appropriate foods available in shops due to food shortages (The Food Foundation, 2021). According to Loopstra (2020), those most at risk of food insecurity during the beginning of the pandemic included Black and Ethnic minority groups (BAME), as well as those who were unemployed, living with children, or had a disability or other health problems; these groups had also been identified as most at risk before COVID (Loopstra, Reeves and Tarasuk, 2019;

Sosenko *et al.*, 2019). The additional layer during the pandemic was now the increased risk for adults who were severely clinically vulnerable to the virus or who were self-isolating (Loopstra, 2020).

At the same time, financial struggle became more widespread during COVID, with the global economic downturn and losses of income from the pandemic continuing even after lockdown (Laborde *et al.*, 2020; Power *et al.*, 2020). As of 2021, it was estimated that 22% of all UK households had lost income since before the pandemic (The Food Foundation, 2021). All these challenges have increased dependence on emergency aid services, with it being estimated that UK foodbank use doubled with the onset of the pandemic (EFRA, 2020a); however, food insecurity, and the use of foodbanks as a means to cope with it, had been rising long before (Boyle and Power, 2021; Pool and Dooris, 2022).

Experiences of food insecurity often coincide or are spurred by other shocks or challenging life situations, such as bereavement, job loss, or relationship breakdown (Garthwaite, 2016). This is seen through what the Trussell Trust foodbank network, the largest in the UK, reported as the main drivers of food insecurity before the pandemic, which include: gaps, reductions, and inadequacy of benefits; challenging life experiences, such as eviction or divorce; ill health, especially mental health; and a lack of informal support from friends and family, often due to exhausted or lacking social networks (Sosenko *et al.*, 2019). Each of these factors was affected by COVID-19, becoming more widespread and amplified for many (Bramley *et al.*, 2021; The Food Foundation, 2021). Indeed, the COVID-19 crisis severely impacted all four dimensions of food security - food supply, food access, and food utilisation; the shock of crisis meant that these dimensions were no longer stable and consistent over time. The ways that the COVID-19 crisis impacted food security dimensions are explored in the following sections, focusing on the UK context.

1.4.3.1 Availability: Food supply issues

The COVID-19 pandemic exposed the drastic vulnerability of the global food supply chain. In the UK, the lack of coordination in the national food supply chain was obvious to consumers who found empty supermarket shelves at the beginning of lockdown. This was influenced by the stockpiling, hoarding, and panic buying behaviour of consumers during this time, as well as the closure of the food service sector with lockdown restrictions, which placed greater demand on supermarkets (Power *et al.*, 2020; Wentworth, 2020; Keane and Neal, 2021).

The inability of the UK food supply chain to rapidly adapt and respond to the shock of the COVID-19 pandemic was influenced by a variety of factors. The supermarket supply chain relies on ‘just-in-time’ logistics based on demand models; when demand in supermarkets surged during the pandemic, there was just not enough buffer supply to cope with this (Garnett, Doherty and Heron, 2020; Wentworth, 2020). Although there was an opportunity to redistribute foods that would have gone to the catering, hospitality, and restaurant sectors to supermarkets and the charity sector, this was often not possible due to lack of appropriate packaging, inability to market certain items, and logistical issues (Petetin, 2020). The inability to cope with shock, adapt to changes in demand, and redistribute foods quickly resulted in an increase in food loss and waste (Aday and Aday, 2020; Petetin, 2020; Filimonau, 2021).

In addition, the lack of diversity in suppliers for supermarkets created a point of vulnerability; issues with one supplier or organisation within the supply chain caused it to become deadlocked (Garnett, Doherty and Heron, 2020). Some countries, such as Russia and Kazakhstan, created temporary export restrictions to ensure their own populations were fed, which thus disrupted global supply chains (Petetin, 2020). Further, the UK is heavily reliant on specific trade routes, with a large proportion of fresh foods, dairy, and meat imported via the Dover Strait ferry network and Channel Tunnel routes; this exposed a single point of failure, exaggerated further by the UK's reliance on imports to maintain food security (Garnett, Doherty and Heron, 2020).

At the same time, the agriculture, food processing, and transport sectors were impacted by severe labour shortages as a result of quarantine measures, international travel restrictions, workplace safety concerns, and COVID-related illness and death (Petetin, 2020; Stephens *et al.*, 2020). One example is the UK horticulture sector, which relies heavily on seasonal migrant workers (Office for National Statistics, 2018). This sector was particularly impacted as there was not enough people available to plant or harvest fruits and vegetables, although this was already an issue prior to COVID due to Brexit (Stephens *et al.*, 2020; Wentworth, 2020; Lin, Lloyd and McCorriston, 2021). Similar issues were also seen within transport and meat processing facilities. In many cases, this led to increased food loss and waste as produce was left unharvested; meat was unable to be processed in abattoirs; and foods rotted at borders during transport (Aday and Aday, 2020; Petetin, 2020). A paradox erupted as people were experiencing empty supermarket shelves, while at the same time, dairy farmers were forced to pour milk down the drain (Aday and Aday, 2020; Yaffe-Bellany and Corkery, 2020; Lin, Lloyd and McCorriston, 2021). We thus see how the national food supply chain lacked the resilience to adapt to the shock of crisis, which left many without food despite there being enough food produced at the time.

1.4.3.2 Accessing food

The issues in agriculture, food processing, and transport sectors transcended through the supply chain and were felt poignantly by consumers as rates of food insecurity grew. The limited options of food available on supermarket shelves, spurred by panic buying behaviours and supply chain issues, meant that affordable brands were usually gone first. This became a concern for vulnerable individuals as well as those struggling financially, as they may not have been able to stockpile and were likely faced with options of only more expensive products (Power *et al.*, 2020).

The lack of available and affordable food items was furthered by new financial and physical barriers to accessing food during the pandemic. The pandemic exacerbated existing economic vulnerability and created new financial hardship for many (Lambie-Mumford, Loopstra and Gordon, 2020; Loopstra, 2020). This was often related to decreases in income due to job loss, lowered wages, decreased hours, or being furloughed, coupled with increases in household expenses (Loopstra, 2020; The Food Foundation, 2021). By August 2020, it was reported that over a fifth of people in the UK had less income than they did before the pandemic (The Food Foundation, 2021).

Physical barriers to accessing food came with the need to self-isolate and with potential illness. Atop this, it became increasingly difficult to obtain food deliveries from shops, even for those shielding who were supposed to have priority delivery slots (Dempsey *et al.*, 2021;

McNeill, Dowler and Shields, 2022). People who could rely on social networks for shopping were able to mitigate this physical access barrier, but this also meant those who were the most socially isolated were left struggling (Blake, 2021). Some were forced to rely upon strangers for shopping and food support, which meant limited food options and thus could be more expensive.

We see that the lack of an adequate food supply, atop affordability and physical access challenges, were direct barriers to accessing food; however, we also see that these were furthered by other determinants which limited the ability to overcome these barriers. The lack of a social support network severely limited options available to people to overcome newfound financial and physical challenges, as had been identified previously in UK food bank research (Sosenko *et al.*, 2019). This suggests that the resources needed to achieve food security go beyond having a locally-available supply of food and the financial capital to afford it.

1.4.3.3 Food utilisation: Links with physical & mental health

To achieve food security, one must also have the physical and mental capability to go to the shop, buy food, prepare it, and eat it. Challenges to these capabilities were threatened by risks to mental and physical health spurred by the pandemic, which were then compounded upon by rising food insecurity.

The recognition of how physical capabilities can inhibit food security, usually hidden in the household, was brought to the forefront during COVID-19. This was shown through the precise classifications of vulnerability to the virus, based upon pre-existing physical health conditions, which informed isolation guidance and thus access to government-delivered food. Prior research, focusing mainly on elderly and disabled people, has also highlighted how strength, the ability to stand at a stove, and the ability to carry food home from shops can impact where people can shop and what they can eat, thus influencing diets and food security (Burns et al., 2011; Lee and Frongillo, 2001; The Food Foundation, 2021). During the pandemic, illness from the virus itself also created an additional barrier. It has also been found that chronic ill-health can influence the ability to achieve food security by affecting the financial resources available (e.g., increased household expenses and impacts on employment options) and by constraining the ability to adopt coping mechanisms to stretch household resources (Heflin, Corcoran and Siefert, 2007; Tarasuk *et al.*, 2013).

The physical capabilities needed to achieve food security are also diminished by the experience of being food insecure, thus creating a vicious and reinforcing cycle. Indeed, Tarasuk *et al.* (2013) suggests that a bi-directional relationship likely exists between food insecurity and chronic ill-health. Food insecurity is often tied to poorer diets due to the combined issues of low access to healthy foods in one's local area and the generally higher cost of healthy foods (Kolodinksy and Cranwell, 2000; Baker *et al.*, 2006; Jetter and Cassady, 2006; Hilmers, Hilmers and Dave, 2012). When faced with financial struggle, food is often one of the first areas where cuts are made, as these costs can be 'squeezed' unlike rent and utility expenses; this often means 'trading down' on the quality of meals or skipping meals entirely (Dowler, Turner and Dobson, 2001; Dowler and Lambie-Mumford, 2015; Blake, 2021; Sheffield City Council, 2021; The Food Foundation, 2021). An increased reliance on food support, which became much more commonplace during the pandemic, also limits food choice and can in some cases result in poorer diets with less fresh fruits and vegetables (Verpy, Smith and Riecks, 2003; Douglas *et al.*, 2015; Middleton *et al.*, 2018).

The poorer diets associated with food insecurity can lead to related health problems, such as diabetes, high blood pressure, and heart and circulation problems (Dowler, Caraher and Lincoln, 2007; Berlant, 2011; Gundersen and Ziliak, 2015). These physical health challenges diminish strength, stamina, and mobility, impacting the physical capability to utilise food (Walker et al., 2010; Wrigley et al., 2003). The poor diets and health challenges associated with food insecurity (e.g., diabetes, obesity, hypertension, and cardiovascular disease) were also found to increase risk to and severity of the COVID-19 virus (Butler and Barrientos, 2020; Holman *et al.*, 2020; Popkin *et al.*, 2020; Srivastava, 2020; Hacker *et al.*, 2021; Merino *et al.*, 2021). Illness from the virus, of course, created a further barrier to accessing and utilising food. One study further predicted that reducing food insecurity and improving access to exercise could have prevented 10,800 COVID-19 fatalities in the U.S. in 2020, prior to the broad availability of vaccines (O'Hara and Ivanic, 2022). This highlights the importance of place (Blake, 2018), where the ability to access and afford healthy foods in one's local area is a major resource to prevent health challenges and ensure the ability to consistently achieve food security.

At the same time that physical health challenges created barriers to food insecurity, and food insecurity created further physical health challenges, mental health was also deteriorating and affecting the future capability to achieve food security. Mental illnesses have long been associated with food insecurity, both as a result and as a driver. Past studies have shown that increased stress and high levels of mental illness, especially anxiety or depression, exist amongst food bank users (Perry *et al.*, 2014; Garthwaite, Collins and Bambra, 2015) or more generally the food insecure (Gundersen and Ziliak, 2015; Martin *et al.*, 2016; Myers, 2020; Pourmotabbed *et al.*, 2020). Martin *et al.* (2016) further suggests a bi-directional relationship where food insecurity can cause poor mental health and poor mental health can cause food insecurity. The risk of mental illness can be increased by higher levels of social isolation and stress, also associated with food insecurity, thus creating another vicious and reinforcing cycle (Martin *et al.*, 2016; Blake, 2019). As mental health deteriorates, this limits the mental capability to budget, plan ahead, and adopt other coping strategies (Heflin, Corcoran and Siefert, 2007; Tarasuk *et al.*, 2013). This is explained simply by a key informant within the UK Trussell Trust food bank network's research report: "If people absolutely haven't had enough to eat, it's very hard to think straight, it's very hard to make good decisions" (Sosenko *et al.*, 2019).

The changes in daily life associated with the COVID-19 pandemic severely impacted people's mental health and thus their capability to achieve food security. Studies showed that wellbeing deteriorated and mental health challenges increased in the UK population during the pandemic as opposed to before, with younger people (age 18-34) and women consistently cited as the groups showing the largest changes (Li and Wang, 2020; Pierce *et al.*, 2020; O'Connor *et al.*, 2021; Office for National Statistics, 2021; Daly, Sutin and Robinson, 2022). Possible reasons for worsening mental health during the pandemic include stress and anxiety associated with getting the virus, newfound financial struggle, increased isolation, and difficulties in managing pre-existing mental illness and accessing mental health support services during lockdown (Cowan, 2020; Holmes *et al.*, 2020).

Rates of loneliness (‘perceived social isolation’) also increased during the first UK lockdown (Groarke *et al.*, 2020; Li and Wang, 2020), which likely negatively affected physical and mental health based on past studies describing these links (Heinrich and Gullone, 2006; Cacioppo, Hawkley and Thisted, 2010; Victor and Yang, 2012; Beutel *et al.*, 2017). Loneliness was less common for people that had a partner, lived with others, and reported higher levels of social support (Groarke *et al.*, 2020; Li and Wang, 2020). Social networks positively impact health through how they provide social support – defined as “the psychological and material resources provided through social interaction” (Long *et al.*, 2022). Social support has thus been highlighted as one of the most important factors to build resilience and cope with stress during times of crisis (Rodriguez-Llanes, Vos and Guha-Sapir, 2013). As lockdown measures, amplified struggle, and mental health challenges increased isolation between people during the pandemic, this thus diminished another critical resource available to people to cope with food insecurity. COVID-19 sadly and starkly brought to light how a diminished set of resources to achieve food security created harsh barriers to living a happy and healthy life – or even to survive the pandemic at all.

1.4.4 Responses to food insecurity during COVID-19

The COVID-19 crisis thus presents a unique opportunity to examine the various strategies that were put in place to cope with crisis within the food system – to see what worked, and what did not. Crises like these are expected to become only more common in the future with climate change and other political disruptions, seen already with Brexit and the Russian-Ukraine conflict, which has spurred the current cost-of-living crisis in the UK; thus, the end of COVID lockdowns has not meant an end of hardship for many (White, 2021). It is thus important to continue to examine how large-scale shocks disrupt food systems, and the ways in which different parts of the system impact each other and expose vulnerability elsewhere (Garnett, Doherty and Heron, 2020). This section thus provides background on the responses and strategies put in place by the Government, voluntary sector, and other local organisations to address rising food insecurity in the UK during COVID-19. These responses are later critiqued in this research, drawing on the lived experiences of food insecurity during a pandemic.

1.4.4.1 UK Government responses

The Government enacted a wide range of policies and programmes to address the issues with food supply, lost income for businesses and employees, and difficulties in accessing food that coincided with the pandemic and lockdown in the UK. While the Government did increase public expenditure dramatically in order to ‘catch’ people, we saw that many still fell through the cracks of government support.

On the supply side, the Government updated a range of policies to overcome logistical challenges in the availability of food (Wentworth, 2020). Competition laws for food retailers were relaxed, allowing for industry collaboration which meant that retailers could share stock level data, distribution centres, and delivery facilities. Support was provided for food businesses to ensure adequate staff levels, and food supply workers were recognised as key workers, which allowed them to keep working through lockdown. Regulations for both freight and retail driver hours were also relaxed to facilitate food deliveries to retailers and increase available delivery slots for individuals. To cope with the loss of migrant labour, the Government and other food and farming groups collaboratively enacted the ‘Pick for Britain’ campaign (https://pickforbritain.org.uk/) to recruit British people into the agricultural workforce; however, it was estimated that less than 20% of applicants actually went to work on farms during 2020 (O'Carroll, 2020; Milbourne and Coulson, 2021). The Government thus began permitting charter flights from Eastern Europe to bring more seasonal workers to the UK (O'Carroll, 2020).

In terms of financial support, loans were provided to support businesses that lost income with lockdown. The Coronavirus Job Retention Scheme was available from the start of lockdown and allowed businesses to furlough employees and keep paying them, with the Government subsidising up to 80% of wages (Francis-Devine, Powell and Clark, 2021). Later in May 2020, financial support was also provided to self-employed individuals whose income had been affected by COVID-19 (e.g., those reliant on income from events) (HM Revenue & Customs, 2022). However, issues were identified with both schemes. Furlough payments were only provided for employees that lost work completely; thus, those with reduced hours had no similar option to make up the lost income (Adam, Miller and Waters, 2020; Loopstra, Reeves and Lambie-Mumford, 2020). Additionally, the delay in the provision of financial support for self-employed people left many financially struggling during this gap, whilst people also found it difficult to determine their eligibility for this support (Blundell and Machin, 2020; Loopstra, Reeves and Lambie-Mumford, 2020).

Time during the first COVID lockdown was thus characterised by a dramatic rise in applications for government benefits programmes, in particular Universal Credit. Universal Credit is one of the most common benefits programmes nationally, supporting the unemployed, those working but with low incomes, or those with sickness or disability (Department for Work and Pensions, 2020). In just the first four weeks of the COVID-19 crisis (March/April 2020), 1.2 million people in Great Britain started a Universal Credit claim; this was approximately a million more than the usual amount of monthly claim starts (Mackley, 2021). Over the course of a year during the pandemic, from March 2020 to March 2021, the number of people claiming Universal credit doubled from 3 million to 6 million (Department for Work & Pensions, 2021).

However, issues with the benefits system at the time left many struggling financially. People found it difficult to access the online system or receive support on their applications, with some being deemed ineligible or experiencing delays (The Food Foundation, 2021; Robertshaw *et al.*, 2022). For those shifting from other forms of income prior to COVID to Universal Credit, this transition was likely a major financial hit. Indeed, Loopstra, Reeves and Lambie-Mumford (2020) estimated that someone aged 25 and over would have gone from bringing home £1260 per month working full time at the national living wage (£8.72 per hour) to £409.89 per month on Universal Credit. This figure takes into account the temporary £20 weekly increase in Universal Credit payments that the Government enacted in April 2020, which lasted for 18-month period. Although this 'uplift' may still not have been enough for many, it was seen to have been important in reducing rates on food insecurity and relieving the pressure on food banks at this time (Bramley *et al.*, 2021). However, other income replacement benefits, such as Jobseeker's Allowance, Employment and Support Allowance (ESA), and Income Support (IS), did not receive a similar increase; reports from the Trussell Trust foodbank network suggest that if this uplift had been provided to other income replacement benefits, it could have reduced the number of emergency food parcels provided by the food banks 2020/21 by around 30% (Bramley *et al.*, 2021).

An additional issue is that Universal Credit applications require a minimum five-week wait period before the first payment; this wait time, already an issue before the pandemic (National Audit Office, 2020), was listed as a key driver of food insecurity (Bramley *et al.*, 2021). During this wait time, it is possible to receive advance payments, but these are subtracted from subsequent benefits payments as debts. We thus see that people are met with the option of either five weeks with no income or being plunged into debt, which could increase hardship later on (Work and Pensions Committee, 2020b). Although other types of debt repayments were suspended during lockdown, debts taken for advance payments could not be paused due to the inflexible nature of the Universal Credit system (Work and Pensions Committee, 2020a). Thus, despite many calls to reduce the five-week wait time and suspend advance debt repayments (Economic Affairs Committee, 2020; Power *et al.*, 2020; The Trussell Trust, 2020b; Trades Union Congress, 2020; Work and Pensions Committee, 2020a, 2020b; Patrick and Lee, 2021), these policies were not updated and continue to present a challenge to people today (The Trussell Trust, 2022a).

The Government also created programmes to provide direct food support to certain groups. For families with children who normally received free school meals, a national scheme was implemented on 31 March 2020 to provide retail vouchers to these families (Lambie-Mumford, Loopstra and Gordon, 2020). However, the rapid set-up of this scheme led to difficulties in receiving or using the vouchers. Many experienced delays or never received vouchers at all, especially in the first few months of the pandemic; in other cases vouchers were not redeemable at local supermarkets and people had to travel to other shops much farther away (The Food Foundation, 2021).

The Government also began a scheme that delivered weekly food parcels to the homes of those shielding (identified as ‘clinically extremely vulnerable’) from late March until late July (Lambie-Mumford, Loopstra and Gordon, 2020; Dempsey *et al.*, 2021). However, these boxes provided food only for the individual classified as ‘clinically extremely vulnerable,’ not for the entire household or for other individuals who may have been self-isolating because of risk from the virus (e.g., elderly people, pregnant women, and many disabled people) (Lambie-Mumford, Loopstra and Gordon, 2020; Dempsey *et al.*, 2021). Even those who had been advised by their doctors to shield may not have been eligible for the Government scheme, while others who were eligible had deliveries stopped early or never received food parcels at all (Dempsey *et al.*, 2021; The Food Foundation, 2021; McNeill, Dowler and Shields, 2022).

Thus, we see that the Government increased funding, adapted policies, and enacted a wide range of new schemes and programmes to ensure that food was available and that people could afford and access it. However, many undoubtedly fell through the cracks of these programmes, and thus had to rely on other forms of support.

1.4.4.2 Voluntary sector and community responses

Where the market failed to provide a consistent supply of food and delivery options, and where the state failed to provide ubiquitous food and financial support for all who needed it, community efforts rose to fill the gap (Caplan, 2020; Bramley *et al.*, 2021; Rivington *et al.*, 2021).

To address the lack of food in supermarkets and the issues with receiving deliveries, many smaller, locally-based farms began increasing their capacity, taking on new customers, and adapting better to online sales (Petetin, 2020; Jones, Krzywoszynska and Damian Maye, 2022; Krzywoszynska, Jones and Maye, 2022). This was seen in the rise of fruit and vegetable boxes across the UK; indeed, between the end of February and mid-April 2020, it was estimated that UK vegetable box sales doubled (Wheeler, 2020). Many of these schemes had waiting lists as demand surged, but approximately 65% of these vegetable box schemes were prioritising key workers, the vulnerable, or people who were isolating (Wheeler, 2020). These schemes were thus crucial in providing locally-grown, nutritious produce to those who were most vulnerable to both the virus and food insecurity. This shows the immense importance of local farmers and food producers in rising to meet the local demand for food. These local food actors were thus able to adapt to crisis and respond quickly to local demand in a way the national food supply chain could not (Jones, Krzywoszynska and Damian Maye, 2022).

The voluntary sector also played an important role in food distribution during the pandemic. Prior to COVID, this sector has largely been the main option for food aid, filling a ‘welfare gap’ left behind in times of austerity (Caplan, 2016; Beck and Gwilym, 2022). Types of food support prior to COVID included a range of formal and informal options, such as food banks, surplus food redistributors, churches, community kitchens and cafés, day centres, school breakfast clubs, and soup kitchens, among others (Dowler and Lambie-Mumford, 2015; Caplan, 2016, 2020). During COVID, many of the physical spaces where one might go for a hot meal, such as day centres and community cafés, were required to close with lockdown. Some of these places were able to adapt to providing food through deliveries, but not all were able to do so (Graven *et al.*, 2021). However, new voluntary groups were also created to ensure that basic essentials were provided to their local communities (Barker and Russell, 2020; The Trussell Trust, 2020c).

The onus of emergency food provision during the pandemic thus landed on the voluntary sector in many cases. The rising levels of demand for food support during COVID created additional burdens upon this sector amid a time of crisis, as resources were being stretched. The Trussell Trust food bank network, the largest in the UK, saw a drastic 89% increase in the need for food parcels during April 2020, compared to the same month in 2019 (The Trussell Trust, 2020b). Nearly 100,000 households across the UK received food from a Trussell Trust food bank for the very first time between April and June 2020 (The Trussell Trust, 2020b). Food banks thus adapted to recruit new volunteers, provide delivery options, and also offer helpline services for referrals and signposting (Caplan, 2020; The Trussell Trust, 2020a). At the same time, many community cafés and soup kitchens also shifted their services to provide emergency food deliveries (Graven *et al.*, 2021). These adapted and expanded services were critical in order meet the needs of those most vulnerable.

Many new, locally-based volunteer groups were also created, including neighbourhood-based mutual aid groups and National Health Service volunteer groups. During the beginning of the pandemic, it was estimated that 4,000 mutual aid groups were started across the UK; these self-organised volunteers helped bring food, medical prescriptions, and other essentials to people who were self-isolating, and in some cases also provided emotional, financial, and information support (Tiratelli and Kaye, 2020; Fernandes-Jesus *et al.*, 2021; Mao, Fernandes-Jesus, *et al.*, 2021). It becomes clear that while the Government did provide support for people at this time, in many cases it was not enough, and voluntary organisations thus rose to fill the gap (Bramley *et al.*, 2021).

Recognising the new burden being placed on the voluntary sector, the Government provided new funding support, seen through grants such as the Coronavirus Emergency and Recovery funds and the COVID-19 Emergency Surplus Food Grant (The Trussell Trust, 2020a). However, the more informal voluntary responses, or those not registered as official charities, may not have been able to access much of the funding available. Despite funding, it became clear that the main assets for voluntary responses at this time were the people who were ready and willing to volunteer, as well as the local resources and partnerships that were made (The Trussell Trust, 2020a; Fernandes-Jesus *et al.*, 2021; Mao, Fernandes-Jesus, *et al.*, 2021; Wakefield, Bowe and Kellezi, 2022). These groups drew upon the people in their communities – the social capital – as well as local community assets to facilitate crisis response.

In Sheffield, England, a wide range of community projects soon became the leaders of the city's emergency food response (Koseda, 2020; Voluntary Action Sheffield, 2020). One example is the Foodhall Project, a relatively small community kitchen and café that became one of the city's largest emergency food providers during the first UK government-imposed lockdown (March-July 2020). This organisation provides an invaluable case study to examine the role of community food projects in responding to the rise of food insecurity during COVID-19, and thus serves as the focal point for the research in Chapter 5 of this thesis.

1.5 Thesis aims, objectives, and outline

The overall aim of this thesis is to explore the sustainability and resiliency of local food systems. An interdisciplinary approach was employed to provide a more holistic view of these concepts. Emphasis was placed on identifying key advantages and disadvantages of local food systems, as well as relationships and trade-offs between different parts of these systems.

1.5.1 Objectives

To meet this aim, this thesis is comprised of three major research projects focused on different aspects of sustainability within a local food system: 1) environmental impacts from food production, processing, and transport; 2) ecosystem service provisioning; and 3) food security. The main objectives and research questions explore in each of these studies (1-3) are as follows:

1. To evaluate environmental impacts from local food production with different scales and management practices.

- How do farm scale, distance to consumer, and management practices influence environmental impacts from local food production?
- What types of local agricultural models result in the lowest environmental impacts per crop output?
- Which factors drive higher or lower environmental impacts across farms?
- Are there trade-offs between different environmental impacts?
- How do the above vary based on geographical context?

2. To assess soil health as an indication of soil-based ecosystem service provisioning across different types of farms and farm landscapes.

- What services and disservices are being provisioned across the farm as a whole, which were not captured within the assessments of environmental impacts (**Study 1**)?
- How does soil health vary both within different landscapes on the farm, and between local farms with different management practices and degrees of urbanisation?
- What are the main factors or variables driving these differences?

3. To understand experiences of food insecurity and the role of community response efforts during a time of crisis.

- What is the lived experience of food insecurity for individuals receiving emergency food support during the first COVID-19 lockdown in the UK?
- What are the main barriers to accessing food during this time?
- How have community responses addressed food insecurity during COVID-19?
- What is the role of these community responses as a component of a more sustainable and resilient food system?

These overarching objectives and research questions are explored within a series of case studies within the U.S. (Study 1) and UK (Study 1-3). Obviously, not all issues of food system sustainability or resiliency can be captured within this thesis, but these case studies provide key insight to multiple elements of the food system, enabling a better understanding of the relationships and trade-offs between different parts.

1.5.2 Thesis outline

The rest of this thesis is structured as follows. In the next chapter (**Chapter 2**), I provide the methodology for each of the three main studies of this thesis. Section 2.1 includes details on the lifecycle assessment method used to evaluate environmental impacts from local food production [**Objective 1**], as well as the lifecycle inventories for all case study farms. Section 2.2 provides detail to the soil health measurements performed across farms in the UK [**Objective 2**]. Finally, Section 2.3 describes the qualitative methods used to understand experiences of food insecurity during COVID-19 [**Objective 3**] and gives further background about the community food organisation evaluated as a case study.

Following this, I present results and discussion for each of these three studies in separate chapters. In **Chapter 3**, I compare environmental impacts from food production across different types of local farms that employ organic and conventional management practices in two case study locations: Georgia, USA and England, UK. In particular, environmental impacts associated with seedling production, crop cultivation , processing / storage, and transport of two widely sold vegetable crops – kale and tomatoes – are evaluated using a lifecycle assessment (LCA) methodology across 11 case study farms in the U.S. and 14 in the UK. Two case study locations were included to allow for an assessment of how geographical and legislative context may play a role in influencing what the 'ideal' model is for a given location. The agricultural models evaluated include: urban organic; peri-urban organic; rural organic; and rural conventional farms.

A scenario-based approach is utilised in this study to evaluate the influence of methodological decisions and assumptions made by the LCA practitioner on the final results, thus providing an objective critique to how outcomes may be shaped within LCA. Three different allocation scenarios are utilised, which are defined for the processes that present the highest uncertainty within literature regarding how burdens should be allocated. Overall, the research showed that environmental impacts varied widely across individual farms, influenced mainly by the specific management practices employed on each farm and the allocation methods used in the LCA, but also by the geographical context and scale. This Chapter concludes with a discussion on the 'ideal' models for local agriculture based on the outcomes of the LCA, whilst also highlighting the limitations of LCA as a method.

In **Chapter 4**, I evaluate key metrics of soil health across a subset of the UK farms included in the prior LCA study to provide context to additional ecosystem services (and disservices) that may be provisioned by these farms. Due to the select number of farms included in this assessment (n=10), this study does not provide wide-sweeping claims about soil health between different farming systems. Rather, potential environmental advantages and disadvantages are highlighted in order to expand the discussion on farm sustainability from the LCA. Soil health is tested across various landscapes on each farm to provide insight to the potential services that may be generated by both cultivated and uncultivated areas. This study revealed that the organically-farmed and uncultivated soils tended to have better soil health than the conventionally-farmed soils, indicating that organic farms, as well as 'unproductive' landscapes on farms (such as grassy fields, woodlands, and hedgerows) are important sources of ecosystem service provisioning. Chapter 4 thus provides a further avenue to critique the sustainability of different local agricultural models, as well as the methods for evaluating them.

In **Chapter 5**, the focus shifts from agricultural production to the consumer to explore another key concept of food system sustainability – food security – and the many factors that influence this beyond just food supply or affordability. This research focuses on the lived experience of food insecurity for those receiving emergency food support during the first COVID-19 lockdown in Sheffield, UK. Emphasis is placed on understanding the barriers to accessing food, the barriers to coping with food insecurity, other aspects of wellbeing (e.g., health and social life), and the role of community responses during this time of crisis. This research provides insight to the multi-dimensional nature of food insecurity and its links to mental health, physical health, and social capital. I conclude that the resources necessary to achieve food security are much more than purely financial and also include other household, social, and place-based resources that enable resiliency in times of crisis.

Finally, in **Chapter 6**, I present a general discussion and conclusion, drawing together the three main research projects to provide a more holistic insight to sustainability and resiliency within a multifunctional food system. I uncover key trade-offs between different sustainable agriculture and food system approaches, giving further context to the overarching research and discourse in the sector, as well as discussing the influence of policy in shaping farm and landscape management decisions. I highlight how the increased role of local governance in food policy can help to build resilience within local food systems through targeted support and partnerships. I conclude that 'more local' food production is not always more resource-efficient or less environmentally-impacting, but that local farms can be key providers of ecosystem and social services to their communities.

1.6 Research timeline and chronology

Figure 1 provides an overview of when different projects and aspects of this thesis research took place during the time of the PhD (October 2018 – January 2023). The initial months of the PhD were dedicated to developing the aims and objectives on which the research would focus. This involved reviewing academic literature about local food systems and urban agriculture to find data and knowledge gaps, as well as meeting with relevant stakeholders to identify research areas of interest for the communities that this research aimed to serve. Informal meetings were arranged with community food organisations and charities, farmer groups, and city officials in Sheffield and Atlanta; these conversations then helped inform which types of studies on local food systems would be most impactful. Based on this scoping, a research plan was developed and submitted in the Confirmation Review report (July 2019).

The first major research project of this PhD was the lifecycle assessment (LCA) study investigating environmental impacts across local farms in Georgia, USA and England, UK, which is indicated in Figure 1 in blue, and results presented in Chapter 3. Lifecycle inventory collection took place through farm visits, calls, and surveys, which was completed in April 2020; thereafter, modelling and analysis for the LCA continued over time until September 2022, with iterative corrections made based on farmer input.

A portion of this PhD research took place during the COVID-19 pandemic, with the first lockdown in the UK beginning in March 2020. During this time, the author of this thesis volunteered with local food distribution efforts in Sheffield, particularly those of a community food group called 'Foodhall.' As the pandemic accentuated issues of household food insecurity, this was a poignant time to investigate these challenges. Although not originally planned within the thesis research, in July 2020 a qualitative research study was thus initiated in collaboration with Foodhall to capture the lived experience of those struggling to access food in Sheffield during the pandemic; this second major research project is indicated in Figure 1 in pink, with results presented in Chapter 5 of this thesis.

The author of this thesis also undertook various work placements during the time of this PhD. These included a fellowship with the Parliamentary Office of Science and Technology (POST) from May-July 2021, as well as a part-time work placement with Foodhall from October 2021-January 2022, which was funded by the University of Sheffield's Postgraduate Research Experience Programme. This part-time work placement was initiated after the research project with Foodhall was complete and focused on helping the community organisation develop a data collection system for impact tracking.

The final major research project of this PhD involved soil sampling across the UK farms that participated in the LCA study, to test for soil health / ecosystem service indicators. Field sampling took place from October-November 2021, and soil samples were tested in the laboratory from March-June 2022. This project is indicated in Figure 1 in green, with results presented in Chapter 4; although the soil study chronologically took place after the qualitative research study, results for the prior are presented in this thesis first to facilitate discussion around environmental impacts and benefits on local farms.

Thesis writing took place from September 2022-January 2023. Introduction material was based upon the initial literature scoping as provided the Confirmation Review, which was then updated and supplemented based on new relevant literature.

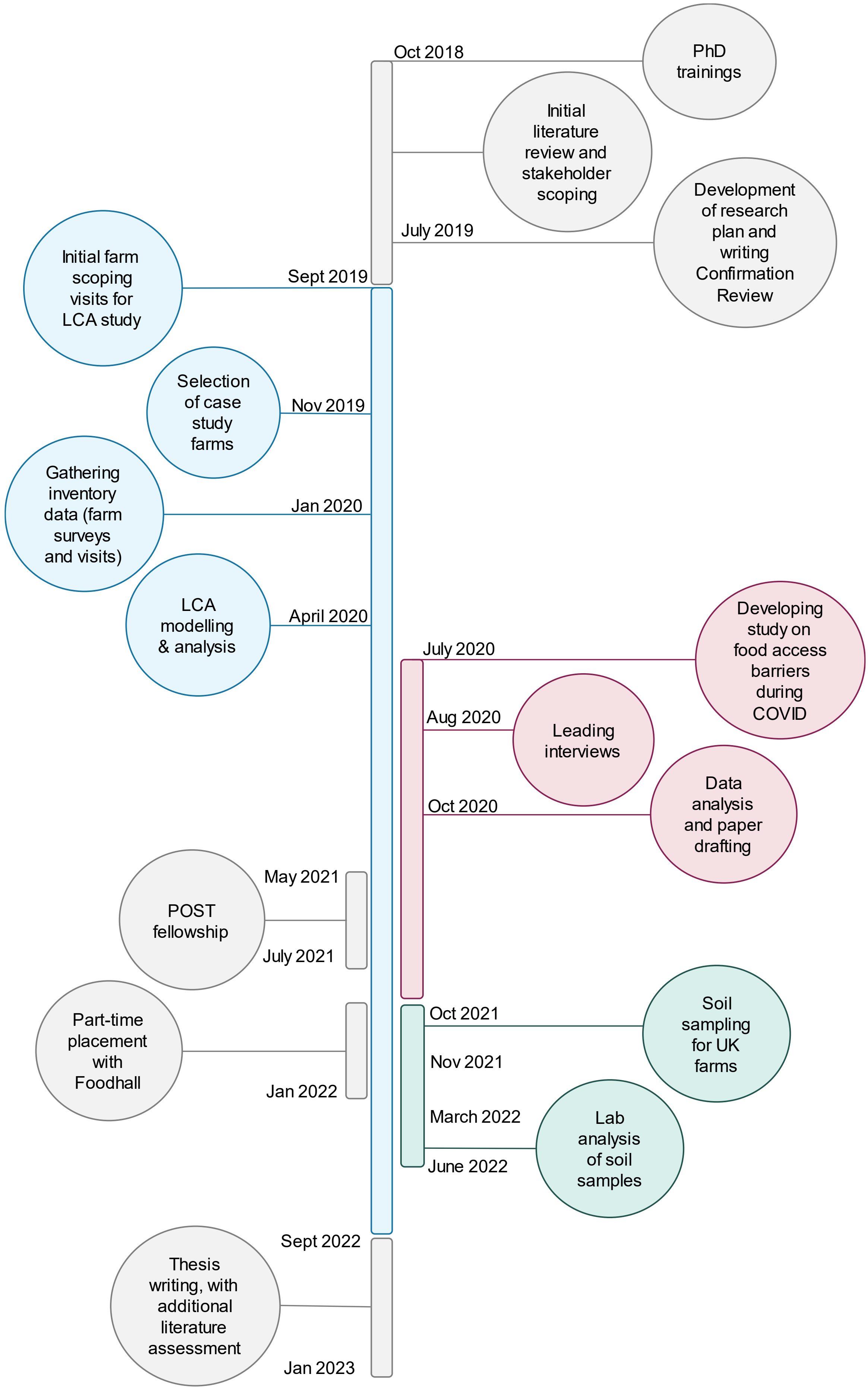

Figure 1 *- PhD research timeline*

1.7 Original contributions of thesis research

This thesis contributes original empirical, methodological, and theoretical advancements to the growing body of research investigating the sustainability and resiliency of agriculture and food systems using an integrated interdisciplinary and systems-level approach. This body of work brings forward new empirical evidence that unravels commonly held assumptions about the sustainability of local agriculture, whilst also critiquing the methods typically used to assess this. This research also highlights the other social and geographical dimensions necessary to achieve food system goals, particularly food security, and provides a new theoretical framework delineating the resources necessary to be food secure and cope with food insecurity within the context of wealthy nations. These contributions are discussed in more detail in the following paragraphs.

The most significant empirical contribution from this thesis research is realised in Chapter 3, which provides an evaluation of environmental impacts from different local farming models using lifecycle assessment (LCA) methodology. Although there have been previous LCA studies in this area, there is still a lack of those investigating urban and peri-urban farming models using real case study datasets (see: Section 1.2.3.2). Therefore, the main contribution of this research is that it provides the largest and most detailed primary lifecycle inventory dataset for different local farm scales (urban, peri-urban, rural) and management types (organic, conventional) to date (to the author's knowledge). Gathering this highly robust dataset, made with contributions from 25 different case study farms, was possible because of the partnerships formed with key stakeholders (farmers and farming groups) and their continuous engagement throughout the research process. The main benefit of this dataset is that its analysis uncovers the high variability in environmental impacts between crops produced by different local farms in the same country or region; this is particularly seen for small-scale, organic farms, which show up to 12x differences in global warming potential per kg crop. This high variability of impacts between organic farms has been lost in prior studies due to the generally low number of case study farms assessed when comparing to more conventional systems. This research thus unravels commonly held public assumptions that 'more local' or 'organic' food is less environmentally-impacting by showing that this is actually highly dependent on local context and individual farm practices. In this way, this research also provides evidence of why it is necessary to develop robust and context-specific primary datasets within food and agriculture and critically analyse these using LCA, especially if intending to provide information to consumers or influence policy decisions.

Further methodological contributions to the development and use of LCA in the food and agriculture industry are made by testing and exploring a wide range of methodological choices, for example in relation to allocation and emission models. LCA results from different methodological scenarios are evaluated and reported in Chapter 3, providing transparency to how different decisions that are made by the LCA practitioner can influence final conclusions. This research finds that the allocation methods used for reclaimed or recycled organic farm inputs significantly impact final study outcomes. This highlights that LCA methodological frameworks must be carefully considered and critically analysed when using this method for decision-making or comparative purposes.

This idea is further emphasised through the research provided in Chapter 4. This chapter provides insight to ecosystem service provisioning on different local farms in England based on field measurements of soil health indicators. Results were analysed and discussed within the context of the LCA study outcomes to contribute to the growing academic critique of the methods and measures used to assess farm sustainability. Outcomes from this study support claims in the academic field that LCA may not be an appropriate way to compare environmental sustainability across farms because of the inability to capture certain environmental benefits that may be generated by some farming systems (see: Section 1.2.3.1). In comparison to the conventional farms tested in this study, this research indeed showed that there are likely many environmental benefits supported by the generally better soil health found on the small-scale, organic farms analysed, which were not captured through the LCA. Although there are various modelling and scoring methods to incorporate certain environmental benefits (e.g., soil health and biodiversity) directly into LCA, their usage is limited, and there is no scientific consensus on the optimal application of these frameworks. Thus, this thesis research provides a novel methodological and empirical contribution to farm sustainability assessments by analysing field-based, location-specific measures of ecosystem services alongside modelled environmental impacts to inform a more holistic discussion of farm sustainability.

This thesis research also incorporates social science methodology in order to provide rich qualitative context to other parts of the local food system besides agricultural production, specifically addressing the viewpoint of consumers (Chapter 5). This is particularly important to be able to understand the ability of the food system to consistently meet key aims, such as ensuring food security (see: Section 1.1.1), during times of shock and stress – or otherwise investigating resiliency. As part of this thesis research occurred during a global pandemic (COVID-19), this was a poignant time to capture the impact of a worldwide crisis on local food systems.

The research provided in Chapter 5 thus explores the important role played by informal community food organisations in local crisis response efforts during COVID-19 and illuminates the lived experience of food insecurity for those who relied on such efforts. Although there were a myriad of studies exploring issues of food supply and access at this time, both in the United Kingdom and across the world (see: Section 1.4.3), the ways that the pandemic impacted individual households and local food structures varied widely, thus indicating the importance of conducting context-specific, place-based research during this time.

This research provided critical empirical contributions in this regard by highlighting the key barriers that individuals in Sheffield were facing in accessing food, which were largely related to difficulties in accessing local and governmental support services, whilst also contributing evidence to the difficulties faced by under-resourced community organisations in supporting such individuals. This research thus made a highly practical contribution as it informed Parliamentary inquiries on COVID-19 and food supply, as well as the Sheffield City Council's food poverty inquiry (see: Section 2.3.4.4).

The qualitative data gathered in this study also provides the foundation for a significant theoretical contribution regarding household food insecurity in wealthy nations. This contribution builds upon existing resource and capability theories (e.g., Sen, 1999a) to provide a new framework that delineates the variety of individual, family, social, and place-based resources that are needed in order to have the capability to be food secure or cope with food insecurity. This framework particularly highlights the interaction of these components, showing the importance of having social capital and community structures in the places where people live. This framework thus lends evidence to how strict ideals of independence and autonomy, as central to neoliberal-ableism, create further barriers to being food secure.

1.8 Positionality statement

As a 27-year old, middle-class, American white woman undertaking food and agricultural research during a PhD programme, I would like to acknowledge the privilege, power, and biases that come with my social, economic, and cultural background. My research focuses on local food systems in the U.S. and UK, and I have worked closely with a wide suite of food actors throughout the supply chain in both countries. I acknowledge the complexities and systemic power dynamics within the food systems in these countries, whilst also acknowledging how both of these countries exert a biased power over the global food system as a result of historic colonial and current geopolitical influence. I strive to approach my research with humility, openness, and a commitment to understanding, particularly aiming to prioritise and amplify the voices of those often overlooked in dominant discourse. I have endeavoured to design my research around the needs of the communities that I work with and highlight the importance of traditional knowledge and local context in my research. I aim for my research to challenge systems of injustice while working towards a more equitable and sustainable food future, while also being mindful of the privilege and power my positionality and role as a researcher holds.

Chapter 2: Methodology

2.1 Evaluating environmental impacts from local agriculture using lifecycle assessment

The first study presented in this thesis uses lifecycle assessment (LCA) methodology to compare different forms of local agriculture, employing organic and conventional management practices, in two case study locations: Georgia, USA and England, UK. The lifecycle assessment (LCA) method was chosen for this study as it allows for production systems to be explored more comprehensively, taking into account impacts and resource use throughout a product's entire lifecycle. LCAs may follow attributional or consequential approaches, with the prior being adopted for this study. This approach was chosen because attributional LCAs focus on actual resource flows to and from a product lifecycle within a set period of time, which allows for specific crop lifecycles to be investigated and compared using data from farmers and suppliers; this is in contrast to consequential LCAs, which describe how resource flows will shift as a consequence of change in demand for, or provision of, the functional unit (Finnveden *et al.*, 2009; European Commission, 2010).

This LCA has been conducted in accordance with the internationally-recognised ISO 14044 methods (ISO, 2006b). This methodology section follows the stages of LCA as set out in the ISO 14044 standards to provide background to how the LCAs in this study are conducted. This includes the four main stages of LCA: goal and scope definition; modelling of lifecycle inventories (LCI); lifecycle impact assessment (LCIA); and interpretation of results.

2.1.1 Goal and scope

This LCA includes two main aims. The first, overarching goal of this LCA is to evaluate how differences in scale, distance to the consumer, and specific management practices on farms influence environmental sustainability for different types of local agriculture. This is investigated by comparing the environmental impacts associated with the production of two vegetable crops (kale and tomatoes) on urban, peri-urban, and rural horticulture farms that utilise either organic or conventional management practices. Thus, to investigate different types of local agricultural farm models, farms are grouped into the following four categories: urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The second aim for this LCA is to determine the specific practices and processes that contribute most to environmental impacts for each type of farm. To facilitate this, the germination, cultivation, processing / storage, and distribution phases for each crop lifecycle were included within this LCA. The specific processes and practices contributing to impacts for individual farms within each phase of production were reported in order to uncover the main drivers of differing environmental impacts between farms.

This study includes farms across two case study sites: the state of Georgia in the United States of America (U.S.) and England in the United Kingdom (UK). Farms from these two Western countries were incorporated to provide a broader insight to how the sustainability of local agriculture (and the definition itself) may vary across geographical context. Although there are many similarities between these two Westernised, high-income countries, they also differ widely in scale as well as agricultural policies, regulations, subsidies, and standard farming practices, which may influence the environmental impacts observed across these farms.

Vegetable crops were selected to be analysed as these constitute the majority of what is grown on urban and peri-urban farms. Specifically, kale and tomatoes were selected as the crop lifecycles to be evaluated to provide insight to two different types of vegetable crops (one leafy green and one fruiting) that require different cultivation practices. Further, these crops represented some of those most commonly grown between the farms wishing to participate in the study, and are also some of the most common vegetable crops consumed in both the U.S. and UK; for example, tomatoes are the second most consumed vegetable in the U.S. (USDA, 2019d). Additionally, brassica production is one of the major home-produced crop markets in the UK, with 91% of the UK cabbage supply being domestically produced in 2018, compared to only 14% of tomatoes (Defra, 2019).

The results provided from this study are aimed toward several audiences. Firstly, this information is useful for an academic audience, as the study aims to provide further research into the sustainability of local agricultural models and urban agriculture in particular. Secondly, the results from this project have been communicated with the participating farmers, and relevant agricultural organisations, to provide detailed information on the environmental impacts of certain practices and processes used across the farms. This will be useful for farmers aiming to understand the main sources of environmental impacts on their farms and any impact trade-offs, so as to identify ways to improve environmental sustainability. Thirdly, although not directly targeted at this audience, the results from this LCA can provide insight for consumers who wish to understand the environmental impacts of purchasing organic or locally-produced foods. Finally, the results may also be referenced and shared with politicians and policy-makers (e.g., through Parliamentary inquires) to provide insight to the trade-offs in environmental impacts of different scales and types of farming, and also to provide information on 'good' or 'best' practices that may be elucidated.

2.1.1.1 Functional unit

The functional unit within an LCA provides a constant factor to be compared across different lifecycles or systems being analysed. The functional unit for this LCA is one kilogram of kale or tomatoes transported to the final point of sale. It is important to note that this is distinct from the amount of crop harvested, as any waste before final transport of the crop is incorporated within the functional unit.

It should also be noted that whilst included case study farms did provide information about crop varieties used, varieties are not held constant within this LCA; all farms used several varieties, and these were not always the same across farms. Varieties were not considered separately within this LCA because the goal of the study is to evaluate the environmental impact of production systems as they operate. Thus, if one farm produces a lower yield because of the production of heirloom varieties, then those production choices should be accounted for as a valid depiction of that type of farming system.

2.1.1.2 System boundaries

This study employs a cradle-to-gate approach, assessing impacts of kale and tomato crops from the moment the seed is planted, up to the point the crop reaches its final point of sale. System boundaries are defined as in Figure 2, with the main processes included within the scope of this LCA being: germination or seedling production; crop cultivation; processing, storage, and packaging operations that may occur on the farm or before transport to distributors; and distribution of the crop to its final point of sale. This is consistent with the Publicly Available Specification (PAS) 2050 for the assessment of carbon footprints of horticultural products, which suggest inclusion of system boundaries up to the point that the product is delivered to the receiving organisation, but not including burdens of the retail environment (Blonk *et al.*, 2010; BSI, 2012).

Thus, processes and burdens specifically not included in this LCA are those associated with: seed production, the retail environment, food consumption and use, and final disposal of food items. For the prior two items, these are not included due to limited data availability to adequately model these burdens. For the latter two items, these are not considered as it is assumed that these processes would be similar between all farming systems, and thus would not influence final comparisons between systems. Section 2.1.7.2 provides further details regarding additional items or processes that are excluded within this LCA, and any specific assumptions made when modelling resource and material flows in the lifecycle inventories.

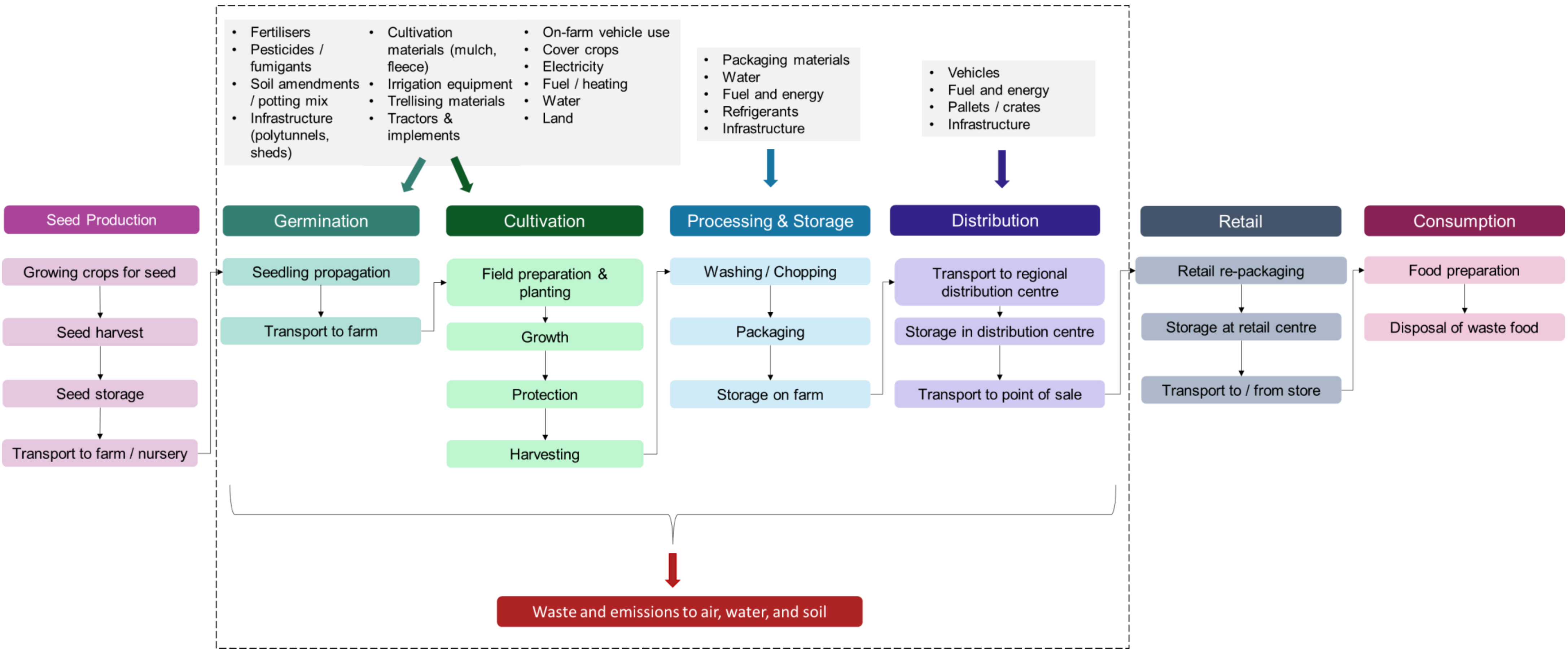

***Figure 2** – LCA system boundaries. Processes within the scope of this LCA are designated by those within the dotted lines, including germination, crop cultivation, processing and storage, and distribution to final point of sale.*

2.1.2 Data collection for lifecycle inventories

Lifecycle inventories are comprised of the input and output flows for a given product system over its lifecycle, quantified in relation to the functional unit (ISO, 2006a). Specifically, this includes any resources and materials required during the product lifecycle, in this case for the kale or tomato crop, as well as any outputs generated during the lifecycle, such as waste and emissions to air, water, and soil.

2.1.2.1 Data sources

In this study, tomato and kale lifecycles are modelled using data provided by 11 case study farm sites in Georgia, U.S. and 14 in England, UK, with further detail on these farm sites provided in Section 2.1.7.1 (Table 1 and Table 2). This ‘primary’ or ‘foreground’ data was collected from farmers via detailed surveys, as well as through site visits and follow-up calls and emails. The information provided by farmers was used to identify the main processes taking place on the farm and to quantify the types and amounts of inputs used, such as fertiliser and pesticides, various materials, energy, fuel, and water. Output flows, such as the amounts of sellable crop and crop waste, were also quantified from this data. Additional information about the make-up of certain items, such as polytunnels, specific fertiliser mixes, and pesticides, was also provided by suppliers, either directly or as gathered through online information and material safety data sheets (MSDS).

Data associated with the ‘background’ system, or processes that take place off the farm, was obtained through secondary databases, as is common among LCA practitioners (De Backer *et al.*, 2009). For example, this includes information about the burdens and emissions associated with the production of inputs used on the farm, such as fertilisers, plastic materials, and machinery, as well as inventories for items with limited available data, such as some types of infrastructure (e.g., sheds). In this LCA, the main secondary database used for this purpose is ecoinvent v.3.7.1, which contains globally-relevant LCI data from a number of sectors, including energy production, transport, and material and chemical production, among others (ecoinvent Centre, 2021). However, in some cases, other databases were used when the ecoinvent database did not have appropriate models for specific inputs used on farms. These included AGRIBALYSE v.3.0 (ADEME, 2020), Agri-footprint v.4.0 (mass allocation) (Blonk Sustainability, 2017), and the World Food LCA Database (WFLDB) v.3.5 (Quantis and Agroscope, 2019), three databases with specific focus on LCIs for the agri-food sector. AGRIBALYSE, although concentrated on food products in France, was useful to provide information about organic fertilisers and machinery commonly used on smaller-scale farms. As each database has its own geographical scope and potentially employs different allocation methods and system boundaries, the modelled processes within these datasets were often used as a basis and then adapted to suit the purpose and methods of this LCA as needed.

Literature data was only used in the case of UK conventional tomato production, as the primary data provided by the case study farm was limited due to the data-protective nature of the controlled environment agriculture industry. All lifecycle inventories were modelled in SimaPro v.9.4 (PRé Sustainability, 2022), which then allowed for the calculation of environmental impacts during the lifecycle assessment phase.

2.1.2.2 Farm recruitment

This section provides an overview of the recruitment of case study farms. Fruit and vegetable farms were contacted both directly and indirectly through various channels. Direct contact with farmers was made either through email or phone call, depending on the contact information provided on websites or directory lists. To make direct contact to farmers, web searches were used to identify farms in various regions of interest that supplied into major cities in each case study location (Georgia, USA and England). In addition, farms were often identified through producer listings for various farmers markets, vegetable box schemes, local supermarkets, and online wholesale / direct sale markets (such as Georgia Locally Grown and Organic North in England). Further, farm directories associated with various organisations were used to identify appropriate farms for the study. In Georgia, the publicly available farm directories associated with the following organisations were used to identify appropriate farms to contact: Georgia Grown (https://www.georgiagrown.com/), Georgia Fruit & Vegetable Growers Association (https://www.gfvga.org/), and Certified Naturally Grown (https://www.cngfarming.org/). In England, farm directories such as the Open Food Network (https://openfoodnetwork.org.uk/), Bristol Food Network (https://www.bristolfoodnetwork.org/), and British Leafy Salads Association (https://www.britishleafysalads.co.uk/) were used.

Indirect contact was also made through advertisement of the study by different associations. In Georgia, cooperative extension offices (https://extension.uga.edu/county-offices.html) and the Georgia Fruit & Vegetable Growers Association helped advertise the study to their farmers and make email introductions. In England, the Brassica Growers Association (http://www.loveyourgreens.co.uk/) and British Tomato Growers Association (https://www.britishtomatoes.co.uk/) also advertised the study.

Farms were screened based on the criteria of 1) being a for-profit, commercial farm and 2) selling produce into urbanised areas within their home country or state. During the recruitment period, over 350 fruit and vegetable farmers in Georgia and 210 in England were screened and contacted. After an initial pool of interested farmers was recruited, preliminary interviews and site visits were conducted to learn more about the main crops produced on each farm, the farmer's management practices, and their markets for sale. At this stage, approximately 21 Georgia farms and 27 English farms were visited or called (in August-November 2019).

After this screening, a total of 11 farms in the U.S. and 14 farms in the UK were then selected to participate in the study. These farms were chosen based on those producing similar crops and to provide a representative number of different farm scales and management practices (e.g., organic vs. conventional). Specifically, in the U.S., this included one urban organic farm, four peri-urban organic farms, three rural organic farms, two conventional tomato farms, and one conventional kale farm. All organic farms produced both tomatoes and kale except for one peri-urban organic farm, which produced only kale. In the UK, this included two urban organic farms, four peri-urban organic farms, two rural organic farms, five conventional kale farms, and one conventional tomato producer. All organic farms again produced both tomatoes and kale except for one urban organic farm, for which kale was not considered due to a particularly bad harvest that year; this thus did not meet the criteria for comparison as an 'average' year. Additionally, for UK conventional tomato production, only one farm was included, but two cases are considered based on different energy sources for glasshouse cultivation: 1) using natural gas heating and the national electricity grid (designated as R-C-NG), and 2) using combined heat and power (designated as R-C-CHP).

2.1.2.3 Farm classification

Farms are classified in this study based on 1) geographical location in reference to urban areas and 2) management practices.

Farms are defined as ‘urban’ (U), ‘peri-urban’ (PU), and ‘rural’ (R) based on urban definitions provided by each country (U.S. and UK). In the U.S., the 2010 Census defines urban areas based on census tracts or blocks which meet minimum population density requirements. The Census Bureau identified two definitions of urban areas: urbanised areas, which contain 50,000 or more people, and urban clusters, which contain at least 2,500 people and less than 50,000 people (U.S. Census Bureau, 2010a). Urbanised areas within this context include both ‘urban’ and ‘peri-urban’ classifications for this study, whilst any area not included within the urbanised area was defined as rural as per the U.S. Census Bureau. ‘Urban’ and ‘peri-urban’ were further differentiated within urbanised areas based on the administrative boundaries of Atlanta, the capital city in the state of Georgia and the area with the highest population densities in the state. ‘Urban’ farms were classified as those within the Atlanta administrative boundaries, while ‘peri-urban’ farms were classified as those outside of the Atlanta administrative boundary, but still within the urbanised metropolitan area as defined by the 2010 U.S. Census.

In England, the Land Cover Map 2015 was used to identify urban, peri-urban, and rural areas (Natural Environment Research Council, 2017). This parcel-based land cover map for the UK was created by classifying satellite data into 21 land cover classes, which were based on the UK Biodiversity Action Plan Broad Habitat definitions (Jackson, 2000). The map identifies urban and suburban land cover classes, which were used to refer to urban and peri-urban areas, respectively. Urban land cover classes include dense urban areas, such as town and city centres, where there is typically little vegetation, as well as docksides, car parks and industrial estates; the suburban land cover classes include suburban areas that have a mix of urban and vegetation spectral signatures (Natural Environment Research Council, 2017). Any area not included within these classes was defined as rural for the basis of farm classification.

Arc GIS was used to overlay farm locations with maps created from the following: the U.S. 2010 Census map, which defined urbanised areas and urban clusters and also provided population densities per census block (U.S. Census Bureau, 2010b); the administrative boundary lines for U.S. states (U.S. Department of the Interior, 2018); the administrative boundary lines for cities in the state of Georgia (Atlanta Regional Commission, 2019); and the Land Cover Map 2015 (25 m raster) for the UK (Rowland *et al.*, 2007). Based on the maps generated, farms were classified as urban, peri-urban, or rural.

Farms were also classified as ‘organic’ (O) and ‘conventional’ (C). Farms classified in this study as organic included: farms that hold organic certification from either the U.S. Department of Agriculture (USDA, 2019e), Soil Association (UK) (Soil Association, 2019), or Organic Farmers & Growers (UK) (Organic Farmers & Growers, 2019); U.S. farms that are certified naturally grown (Certified Naturally Grown, 2023); U.S. and UK farms that are Demeter-certified biodynamic or organic (Biodynamic Association, 2019; Demeter

Association, 2019); and farms that follow organic guidelines for their country and self-identify as organic, but are not able to purchase certifications due to lack of funds. Farms were classified as conventional if they did not meet organic guidelines for their country and did not have any certifications included in the organic classification.

2.1.2.4 Data collection: Surveys and site visits

Participating farmers were then provided questionnaires via email that asked about their tomato and kale production in 2019. The questionnaires included detailed questions on yields and waste amounts, resource use, and energy use throughout various stages of the crop's lifecycle, specifically the nursery, cultivation, processing and storage, and transport / distribution stages. Farmers provided information on specific amounts of the following materials and resources used: water, land, energy, fuel, germination materials (e.g., germination trays, potting media, seeds), infrastructure (e.g., polytunnels, greenhouses, sheds), cultivation equipment and materials (e.g., machinery, irrigation, trellising systems, plastic or natural mulches, harvesting equipment), fertilisers, pesticides, and packaging materials. In addition, farmers were asked to specify periods of use and lifetimes for each resource and material used where appropriate, and this information was then used for allocation purposes.

After reviewing the returned questionnaires, each farm site was visited at least once to gain further information on growing practices, evaluate farm infrastructure, and to weigh and measure various materials used. Follow-up questions were then asked through email or phone calls. As much as possible, information on specific brands and suppliers for each input was recorded so that detailed models for each input could be constructed. Various nurseries and agricultural suppliers used by the farms were also contacted to provide detailed information on materials and production practices. For example, polytunnel suppliers were often able to provide information on types and weights of parts in the structures used on specific farms. Information on specific ingredients in fertilisers and pesticides was also gained through material safety data sheets (MSDS) based on the specific products used by each farm. Collectively, this information was then used to construct detailed and site-specific lifecycle inventories for each crop on each farm.

To maintain confidentiality, all farm names have been kept anonymous in this study. Written consent was recorded at the commencement of initial site visits (before questionnaires were sent) or via email. This study was reviewed and approved by the University of Sheffield Research Ethics Committee.

2.1.3 Impact assessment methods

Following the creation of lifecycle inventories, lifecycle impact assessment (LCIA) is then used to relate elementary flows within the inventories to a set of environmental impact scores, such as for global warming, eutrophication, and acidification, among others (Hauschild and Huijbregts, 2015). During the characterization step of an LCIA, elementary flows are multiplied by characterization factors to translate the flow into an equivalent substance unit (e.g., translating methane into carbon dioxide equivalent) (Huijbregts *et al.*, 2017). This is done at both midpoint and endpoint levels. Midpoint-level characterization factors are located along the impact pathway, usually at the point where the environmental mechanism (impact) becomes the same for all flows in a particular category (e.g., when several emissions or resource flows create a similar impact, like global warming) (Goedkoop *et al.*, 2009). On the other hand, endpoint-level characterization factors collate these midpoint categories into final damage areas, such as human health, ecosystem quality, and resource scarcity. Thus, midpoint characterization factors are strongly related to the inventory environmental flows and have relatively low uncertainty, whereas endpoint factors provide better information on the relevance of environmental flows to final environmental and health outcomes, but have more uncertainty than midpoint factors.

In this study, lifecycle impact assessment was performed using the ReCiPe 2016 v.1.05 characterization method (Huijbregts *et al.*, 2017), with calculations performed in SimaPro v.9.4 (PRé Sustainability, 2022). Figure 3 provides an overview of the lifecycle impact assessment process utilising the ReCiPe 2016 methodology. ReCiPe is the enriched and harmonised successor of the other commonly used Eco-Indicator 99 and CML 2000 characterization methods. It includes eighteen total midpoint impact categories that relate to three endpoint impact categories, namely human health, ecosystems, and resource availability. The ReCiPe 2016 method was selected for this study due to the broad and comprehensive set of midpoint impact categories used in the method; its recent update and thus timely relevance in comparison to other methods; its global scope; and its widespread use among the scientific community (Aymard and Botta-Genoulaz, 2017). In terms of result calculation and interpretation, midpoint impact categories are utilised, as these provide stronger insight to specific environmental damage pathways and have less uncertainty.

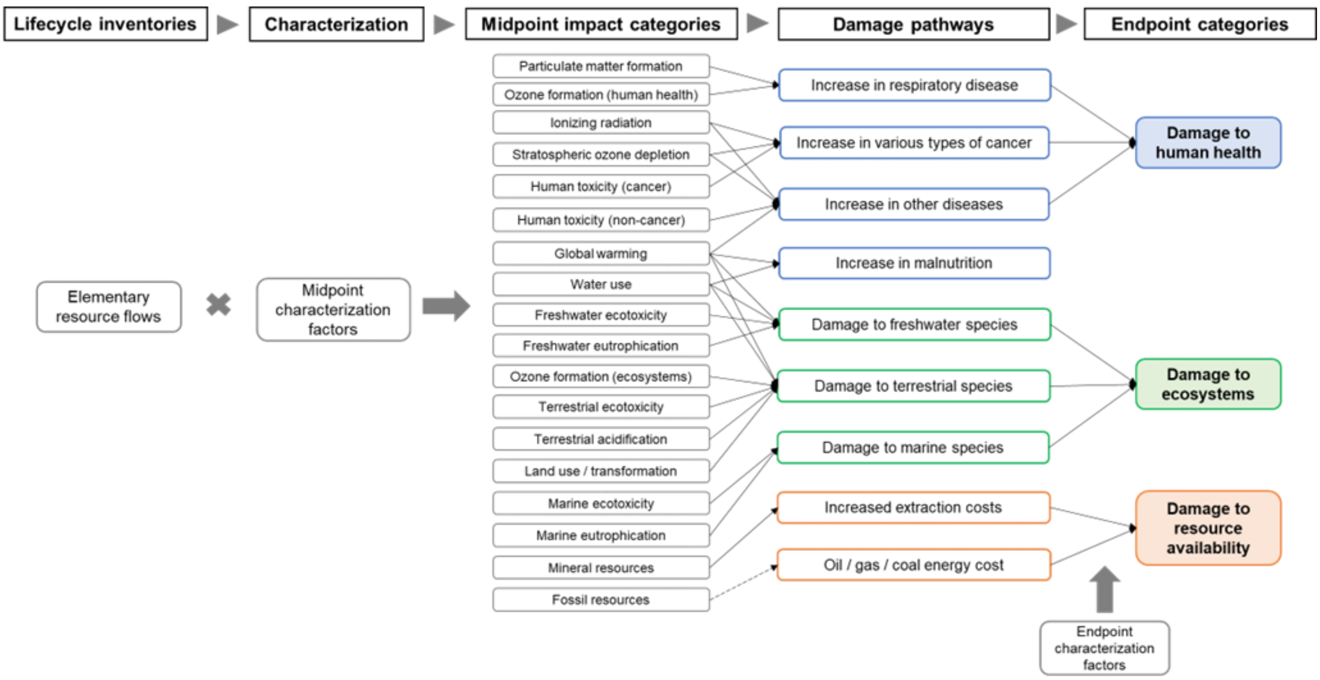

Figure 3 *– Overview of lifecycle impact assessment process based on ReCiPe 2016 methodology, adapted from Huijbregts et al. (2017). The dotted line implies that there is no constant midpoint-to-endpoint factor for fossil resources in the ReCiPe methodology.*

ReCiPe characterization factors are defined for three cultural perspectives, which differ based on their assumptions and sources of uncertainty. The differences mainly relate to choices on issues such as time perspective and assumptions regarding the expectation that future technological advances or policy development will mitigate environmental damages. Specifically, the three cultural perspectives include:

- Individualistic (I) perspective, which is based on short-term interest, undisputed impact types, and technological optimism.
- Hierarchist (H) perspective, which is based on scientific consensus with regard to the time frame, most common policy principles, and plausibility of impact mechanisms.
- Egalitarian (E) perspective, which is the most precautionary and takes into account the longest timeframe and all possible impact pathways for which data is available.

The Hierarchist perspective has been utilised in this impact assessment, as it is based upon scientific consensus and is often considered the method's default, thus providing the most applicable and comparable method for this study (Dekker *et al.*, 2019). This perspective bases global warming potential on a 100-year time frame, which is commonly used among other LCAs. Note that this method assumes global warming potentials of dinitrogen monoxide as 298 kg CO_2 eq (kg N_2O)$^{-1}$ and methane as 34 kg CO_2 eq (kg CH_4)$^{-1}$.

Seven out of the eighteen midpoint impact categories have been selected as the main focus within this LCA, which include: global warming potential (kg CO_2 eq), fine particulate matter (PM) formation (kg PM2.5 eq), terrestrial acidification (kg SO_2 eq), freshwater eutrophication (kg P eq), marine eutrophication (kg N eq), water consumption (m^3), and land use (m^2a crop eq). These have been selected to provide a holistic understanding of some of the main environmental challenges that are simultaneously being driven by agriculture and threatening future food production. All result graphs for these impact categories have been produced using Origin(Pro) v.2022b (OriginLab, 2022).

2.1.4 Basic allocation methods

When considering complex and multifunctional systems such as agriculture within an LCA, decisions must be made regarding how to allocate environmental burdens between the different products in the system. This is particularly important for situations such as co-production, where a process produces several products at the same time (e.g., combined heat and power units); combined production, where several products are produced during a certain period of time, in processes that are dependent on each other (e.g., crop rotations); and recycling, where the waste from one process provides the raw material input for another process (Blonk *et al.*, 2010).

ISO 14044 provides standards for how environmental burdens from different processes should be allocated in these cases (ISO, 2006b). These include (in order of preference): dividing the process into sub-processes, such that no co-products are produced, and allocation is avoided entirely; expanding system boundaries to include co-products, multifunctionality, and alternative production methods; and allocating the environmental burden between co-products according to physical (e.g., mass or area), economic, or other relationships (e.g., energy content).

The first option (avoiding allocation) is often not possible, if the processes can truly not be divided into separate processes; for example, with combined heat and power, which simultaneously produces heat and electricity. The second option, system expansion, therefore becomes ideal; however, inclusion of various co-products throughout the food supply chain would require a very expansive LCA with massive amounts of data collection, when the focus is often on a specific crop, as in this LCA. Thus, in these cases, a simplified method of system expansion, in the form of substitution, is often used to account for other co-products and functions. This simplified method is adopted within this LCA for the cases of compost and combined heat and power production, as one scenario option (see: Section 2.1.5). Otherwise, the third option of allocation, using physical or economic relationships, is often employed; this is used as the default within this LCA.

2.1.4.1 Allocation for multifunctional systems

Agricultural systems are inherently complex and multifunctional, as often many products are produced on one farm using the same equipment and infrastructure (also called combined production). In this LCA, mass, area, or time-of-use is used to allocate burdens to specific crops in the case of combined production. With mass allocation, the resource flows or burdens are attributed to one crop based upon the total mass of the crop produced, out of the total mass of all crops produced that relate to the item or operation in question. For area allocation, flows are allocated based on the area used by the crop of interest, out of the total area that is served by that particular item or process. Other materials or flows may be further allocated based on the time that an item is used for a particular crop lifecycle, out of its total time of use in the year and total lifetime.

The allocation method used within this LCA depends on which is more suitable for each particular context and process. For example, when growers give information about total packhouse operations (e.g., total energy use for all operations over the year), this has been allocated to the crop using mass allocation (percent by weight out of all crops produced), as theoretically a larger amount of produce would require more energy for processing, storage, and packaging. However, if growers give specific information about monthly energy use for a cool room used for crop storage, then this is allocated to the crop of interest based on the area that the crop takes up in that cool room and how long it is stored there compared to other crops.

Thus, materials and flows are also allocated based on the time they are used for a specific crop, often used in conjunction with mass and area allocation. As another example, polytunnel infrastructure is allocated to the crop of interest based on the area that crop takes up in the whole polytunnel and the time out of the year that polytunnel is used for that crop's production. In some cases, however, time-of-use allocation may be used on its own. For example, the use of a germination tray will be allocated to the crop of interest based on how long it is used for that crop's seedlings, out of how long it is used throughout the year and its total lifetime.

Economic allocation is another commonly used allocation method in agricultural and food LCAs, where burdens are allocated to one food item based upon its monetary value compared to other co-products (Williams, Audsley and Sandars, 2006; Ardente and Cellura, 2012). However, this method is not employed within this LCA, as the goal of the study was to focus on resource use, physical flows, and environmental sustainability of the operations, not economic viability. Further, this allocation method is identified as a 'last resort' in the ISO 14044 standard (ISO, 2006b).

2.1.4.2 Cover crops and crop rotations

Many farms in this study use cover crops to improve soil health and to fixate nitrogen. Cover crops are generally not grown to be sold, but instead grown for the purpose of protecting and improving the soil in between cycles of growing other 'cash crops' that are actually sold (Wallander *et al.*, 2021). Cover crops are grown anywhere from a few weeks to several months or years, and then are usually incorporated into the soil to build organic matter and provide nutrients, serving as a 'green manure' (Kumar *et al.*, 2014; AHDB, 2015). Cover crops can also provide a break in pest and disease cycles, decrease weed pressure, and reduce soil erosion and compaction, especially when growing deep-rooting crops (Blanco-Canqui and Ruis, 2020; Wallander *et al.*, 2021). Leguminous cover crops fix nitrogen in the soil, and thus can be a major part of the farm's fertility regime (Dabney *et al.*, 2010). Cover crops are therefore usually employed as part of a crop rotation (Reeves, 1994), which is the practice of growing specific crops in a planned sequence on the same area of land in order to maximise nutrient benefits and reduce pest, disease, and weed pressure (Bullock, 1992).

Within this study, all growers that utilise cover crops have specified their main purpose for doing so – whether it is used as a major source of fertility for the crop of interest, or to provide soil benefits more generally to many crops throughout a rotation. This has been taken into consideration when deciding how to allocate cover crop burdens.

The longer-term cover crops used by farmers in this study are grown from six months to two years within a long crop rotation cycle of four to eight years. These cover crops will provide soil benefits and nutrients for other cash crops throughout the rotation, and thus the burdens of growing these longer-term cover crops must be allocated to all following cash crops within the planned rotation. In these cases, cover crop burdens and resource flows have been allocated to the crop of interest based on the time that that crop is grown, out of the total time that the other cash crops within the crop rotation are grown. For example, if a two-year cover crop is grown within a six-year crop rotation, and the crop of interest is grown within that rotation for six months, then the allocation would be 0.5 years out of four years (12.5%). This allocation is then used to split the burdens of the cover crops between all cash crops that benefit in the rotation.

Other farms in this study also use short-term cover crops, grown for only a few months. These cover crops are employed as 'catch crops' that are used to provide soil cover in between close periods of growing other cash crops and also to take up excess N in the soil to reduce leaching (Dabney *et al.*, 2010). They are generally incorporated into the soil or left on the soil as a mulch to protect the soil and provide nutrients for the following crop. The burdens of growing these shorter-term cover crops have mainly been allocated just to the following cash crop, as farmers have generally specified that these are used to benefit a specific crop. This is in accordance with methodology used within the ecoinvent database (Nemecek and Kagi, 2007).

2.1.4.3 Recycled materials: cut-off allocation

In this LCA, the cut-off allocation method (also known as the recycled content method) is generally used to attribute burdens associated with waste, recycling, and reused materials. Recycling can be considered as a process where a material that would have otherwise gone to waste is instead used as the raw material input to create a new product. If the same material is recycled over and over again, then a (theoretically) infinite number of lifecycles can be considered for the original material. The issue then comes with how to attribute burdens associated with the original production of the material (e.g., raw material extraction) and with the recycling process. This depends on where the system boundaries are drawn around the various lifecycles in a cascading or open-loop recycling system.

In cut-off allocation, environmental burdens are assigned to the product system that directly causes them (Ekvall and Tillman, 1997). Thus, any subsequent lifecycle or function of the product is disregarded in relation to the lifecycle being studied. In terms of waste management, this means that the burdens associated with recycling or repurposing a material will be attributed to the lifecycle of that recycled product (the ‘secondary’ lifecycle), not to the primary or original product lifecycle (Ekvall and Tillman, 1997; Kousemaker, Jonker and Vakis, 2021). However, if recycling or repurposing does not occur, then the impacts of waste management (e.g., landfilling or incineration) would be attributed to the primary lifecycle, or to the last lifecycle that occurs before final waste treatment (Ekvall and Tillman, 1997).

An example of this method will be given in the case of a plastic water bottle (primary lifecycle) that is recycled into a textile (secondary lifecycle). Using cut-off allocation, the burdens of producing the plastic and processing the water bottle are attributed to the primary lifecycle (the water bottle). Once this water bottle is sent to the recycling centre, the burdens associated with recycling the plastic and forming it into a textile would be fully attributed to the secondary lifecycle, or that of the textile. Thus, no waste-related burdens are attributed to the primary lifecycle. If the textile is then brought to a landfill at the end of its life, the burdens of that waste management would be attributed to the lifecycle of the textile as well.

In this way, cut-off allocation is used throughout this LCA to attribute burdens of any recycled materials. Thus, if a farmer recycled the plastic mulch that is used on their farm, the burdens of this recycling process are not counted. However, if the farmer sends this plastic to a landfill, the burdens of that waste disposal are attributed to the crop lifecycle. This method thus favours recycling as a waste treatment process.

2.1.4.4 Reclaimed and reused materials

Another topic of consideration for allocation in LCAs is using reclaimed materials (‘reuse’). This refers to materials that may have otherwise been waste, including by-products, but instead have been ‘rescued’ and given a second life or used for another purpose. This is generally similar to recycled materials, but usually reclaimed materials are not re-processed and thus there is not a ‘recycling process’ to consider.

This LCA makes a distinction between reclaimed materials that are free versus those that the farmer has paid for. It is assumed that items given for free would have been otherwise wasted, and thus these should thus be considered as recycled materials, following the cut-off allocation procedure. However, items that are purchased, even if resold or normally considered a by-product, still have value. Thus, upstream impacts of production are attributed to these materials.

Some examples of reclaimed materials used by farmers within this study include: old polytunnels that may have been gifted from other farms; tyres from mechanic’s shops that are used to hold down plastic netting over crops; and trampoline arcs used to build polytunnels. In these cases, the reclaimed materials were all free and would have otherwise gone to waste had the farmer not claimed them. Thus, these items can be considered as recycled materials, and the cut-off allocation method is applied (Ekvall and Tillman, 1997). Using this method, the upstream burdens of producing these materials are not attributed to their reuse, as they should be attributed to the material's original use. For example, the burdens of producing the tyres should be attributed to the lifecycle of the car, not to their use on the farm. However, the burdens of reusing the materials (seen as the material's secondary lifecycle) are attributed. This includes the burdens of any processes performed to modify the materials once they have been reclaimed, any burdens associated with their reuse, including transport of the items, and the burdens associated with their final end-of-life waste.

For other reused items that the farmer has paid for, the primary lifecycle of the product is still attributed; for example, this could include old tractors or equipment that have been purchased from other farms. However, when attributing the burdens associated with the material's end-of-life waste, this has been considered based on the material's extended lifetime (e.g., total lifetime of use). This is in accordance with the British Standards Institution's specification for the assessment of lifecycle greenhouse gas emissions of goods and services (BSI, 2011).

For natural materials that are 'reused', the case is slightly different. Natural materials used by farmers in this study include items such as leaves and waste woodchips, provided for free by tree surgeons to be used as mulches. For these materials, only the burdens of transport are attributed. Any emissions associated with the decomposition of these materials is not included, as it could be assumed that these materials would have otherwise decomposed elsewhere and produced similar emissions. This rule is also followed generally for fresh manures used on farms as a fertiliser input, although the emissions associated with composting manure are counted. Allocation procedures for manures are discussed in more detail in Section 2.1.5.1.3. For other natural 'waste' materials and by-products that are used by farmers but are not given for free, such as seed meals from oil processing, burdens have been attributed following normal cut-off procedures. Upstream impacts between by-products are allocated based on mass allocation methods.

However, there are still a few instances where allocation is not as clear because of the difficulty in defining system boundaries for certain items and processes. In these cases, several allocation scenarios will be explored within this LCA to see how they influence final results. These particular cases are related to: the allocation of composting and manure burdens (in relation to using waste resources as an input), and the allocation between heat and electricity from combined heat and power units (CHP). These scenarios will be explained in detail in the following sections.

2.1.5 Allocation scenarios

Three main scenarios are explored within this LCA, which differ in methods of allocation for specific processes. Outside of these specific processes (described in the following paragraphs), the cut-off allocation and physical allocation methods are still used for all other processes throughout all scenarios. Thus, the 'default' allocation methods are as follows. Using cut-off allocation, recycling burdens are attributed to the lifecycle that uses the recycled material as an input. Reused products (by-products and other products that would have otherwise been wasted) are not attributed any upstream burdens from the processes that created them. Burdens between co-products are allocated based on physical relationships (generally mass allocation).

However, there are some cases where the application of these allocation methods is not as clear; where varying allocation methods have been used for these cases in literature; and where impact assessment results have been found to be very sensitive to the allocation methods used. This has been identified for the cases of composting, manure storage, and combined heat and power (CHP). The difficulty in defining allocation rules for compost is due to the uncertainty about whether compost is a waste product (by-product of waste management) or a valuable, recycled product. In addition, there no consensus on how to allocate between the co-products of heat and electricity for CHP used in agricultural greenhouses. For these reasons, the allocation of these processes has been explored to elucidate how varying allocation methods may affect final results in agricultural LCAs.

Three different allocation scenarios (numbered 1-3) have been explored throughout this LCA to attribute burdens within these specific processes. Within these three scenarios, only allocation methods for these specific processes are changed; for all other processes, the default allocation rules apply (cut-off and physical allocation). The first two scenarios employ different cut-off allocations for compost and manure, which depend on where the system boundaries are drawn, using the same energy allocation for CHP. The third scenario attempts to avoid allocation all together by using the substitution method of system expansion, where avoided burdens are subtracted from particular processes. The reasoning, justification, assumptions, and specific system boundaries behind these three selected scenarios are explained in detail in the following sections.

In summary:

- Scenario 1): Cut-off allocation for compost as a recycled product and energy allocation for CHP
    - Compost is viewed as a valuable material, made from the recycling and repurposing of green waste. Thus, burdens of the composting process are allocated to the crop cycle in which compost is applied as a fertiliser or soil input.
    - Manure is viewed as a valuable ingredient in the compost. Thus, burdens from composting the manure are included, using the same emission factors for composting of biowaste, for crop cycles which use this manure as an input.
    - Heat and electricity are co-products of CHP. Allocation between the co-products is based on their energy content.
- Scenario 2): Cut-off allocation for compost as a waste management process and energy allocation for combined heat and power
    - Compost is viewed as a by-product created from the waste management process for organic waste materials. Thus, the burdens of the composting process are allocated to the crop process that creates the organic waste, not the crop process which uses the compost input.
    - Manure is a waste product of the animal lifecycle. No burdens of composting manure are included.
    - Heat and electricity are co-products of CHP. Allocation between the co-products is based on their energy content (same as Scenario 1).
- Scenario 3): System expansion with substitution for compost and CHP
    - Composting is viewed as an alternative waste management solution to the municipal solid waste stream. Thus, the burdens of the municipal solid waste

stream are subtracted from composting burdens for the biowaste components of compost only (not manure). This applies only to compost that is used as a fertiliser or soil input for a specific crop process. Burdens from composting of the green waste produced by a crop process are not included (to avoid double counting).

- Burdens associated with manure are the same as in Scenario 2. No burdens from composting manure to be used as a fertiliser input for a crop are included. No avoided burdens are subtracted for the manure components of compost. This is because the alternative scenario for manure management (composting or long-term storage on a livestock farm) likely generates equivalent emissions to composting or storage on a vegetable farm.
- Using CHP, electricity is produced in excess of demand in crop greenhouse production. Thus, burdens of the national grid supply of surplus electricity are subtracted from the burdens to produce that electricity through CHP.

2.1.5.1 Composting allocation

Compost is the result of the controlled decomposition of organic matter under aerobic conditions. It is often used as a supplement to improve soil health, or directly as a fertiliser, especially on organic farms. In this study, compost is both an input and an output of the crop lifecycle. Compost may be made on the farm from various green waste and food waste (e.g., crop residues and waste crop), and is thus seen as a method of processing the waste produced by a certain crop's lifecycle. However, many organic farms in this study also either buy in compost, or make compost themselves, to use as a fertility input for the crop lifecycle.

Compost is thus an important component of organic crop lifecycles, and the results of this LCA are highly sensitive to how the burdens of the composting process are allocated. For this reason, different methods of compost allocation have been explored throughout this LCA, which will be described in greater detail in this section. Note that the different allocation procedures and scenarios discussed in this section refer only to the burdens from the actual composting process (creation of compost from organic waste); burdens from application of compost (i.e., its use as a fertiliser) are applied throughout all scenarios.

Compost can be seen as a component of a cascading system, ora system that results in the sequential reuse of a given material (Rehberger and Hiete, 2020). Green waste and food waste are used to create compost. This compost is then used as an input on a farm to grow crops. These crops generate more green waste, which may again be made into compost to be used as an input in another crop lifecycle. Thus, there becomes an infinite number of crop lifecycles, where compost is both and input and output of the system (not all necessarily produced on the farm). This cascade is shown in Figure 4.

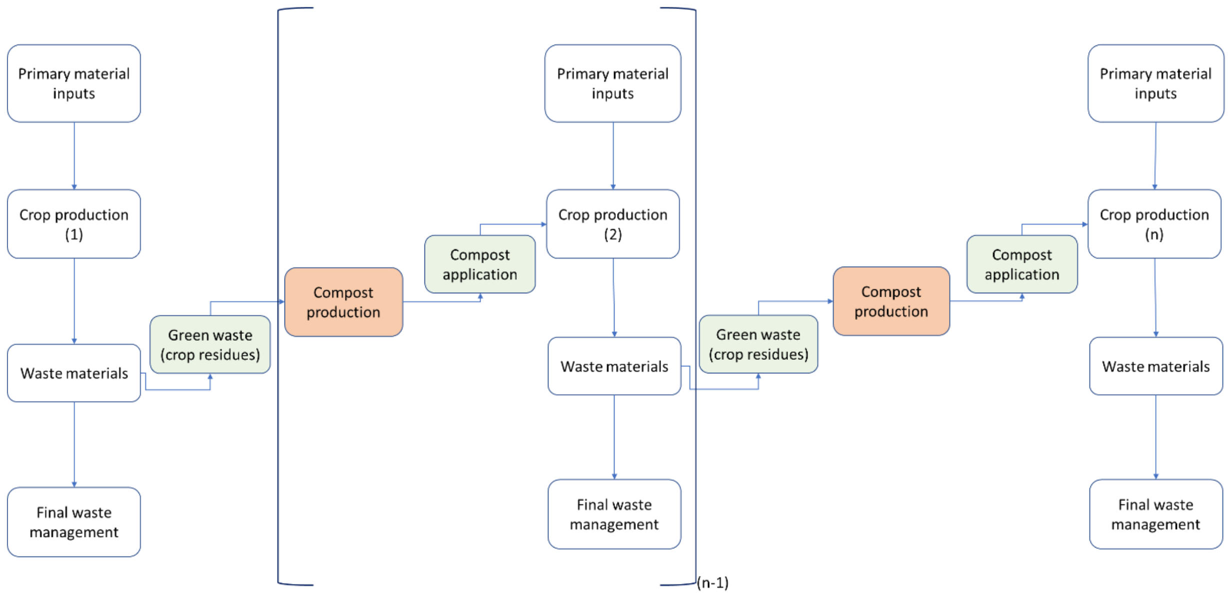

Figure 4 *– Cascade of compost lifecycles, where compost production is both an input to and output from crop production*

Ideally, cascades would be investigated on a system level, defining and attributing the environmental impact of all items in the cascade (Ekvall and Tillman, 1997). However, there is often not enough information available about all other lifecycles in the cascade, and thus boundaries must be introduced around the particular product lifecycle of interest in the cascade (Tillman *et al.*, 1994). This is necessary to attribute compost as a farm input or output at any one point in the crop lifecycle, as there is not sufficient information about the prior or later lifecycles of the compost or organic material.

2.1.5.1.1 Overview of composting allocation options used in literature

When compost is used as an input or produced as an output on a farm, the main issue is where to assign the burdens of the composting process. This includes the transport impacts of the organic material, energy, fuel, materials, equipment, and infrastructure used to produce the compost from organic waste, as well as the fugitive emissions that come from the compost pile, which are discussed in detail in Section 2.1.7.6.4. The composting of organic waste is a unique process because it can be viewed as both a ‘recycling’ process that creates a new and valuable product, or simply as a waste management strategy that produces a by-product. How compost is viewed and how composting burdens are allocated is done very differently throughout agricultural LCA studies; there is no standard method, and the attribution of impacts relies on the method chosen by the LCA practitioner. Figure 5 provides a summary of the main allocation strategies used for compost throughout LCA studies, including both cut-off allocation methods and system expansion methods.

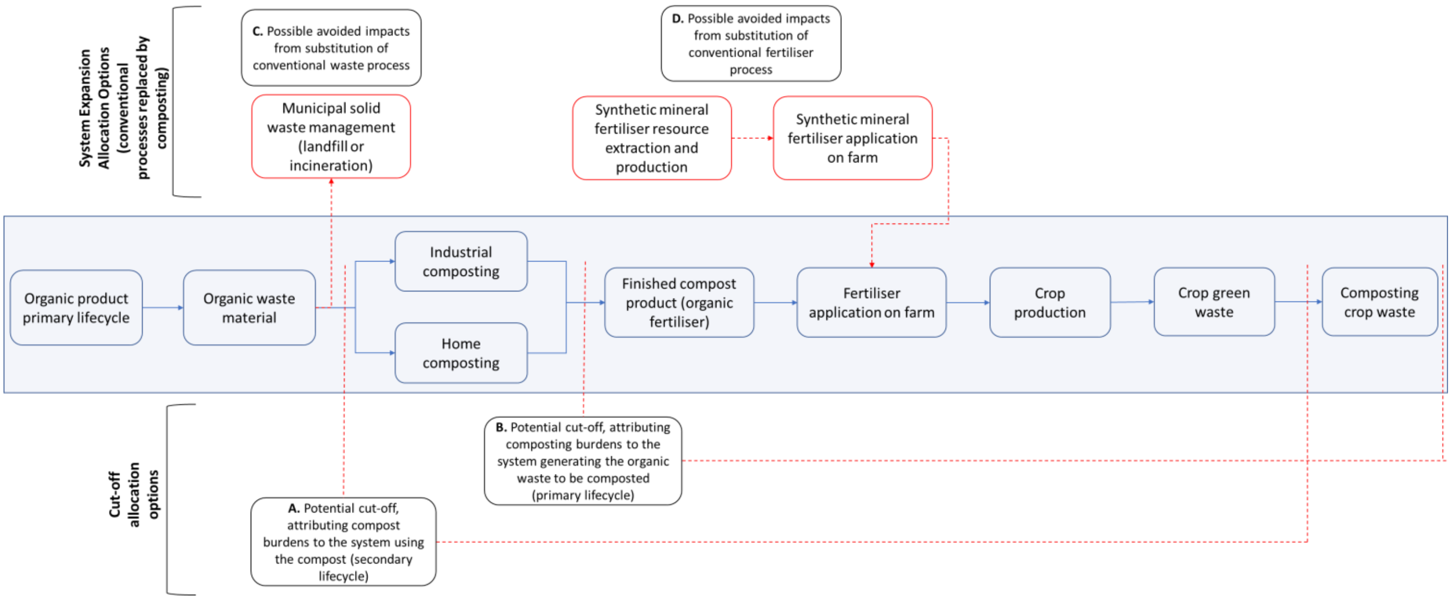

Figure 5 *– Literature allocation options for burdens associated with the composting process. This includes options for cut-off allocation (A., attributing composting burdens to the system applying the compost, or B., attributing composting burdens to the system generating the organic waste) and for system expansion to include conventional processes which would be substituted by composting (C., municipal solid waste management or D., conventional fertiliser production and application).*

Throughout the rest of this LCA, the recycled-content cut-off allocation method has been used. As previously discussed, this approach allocates environmental burdens to the 'recycled' or 'reused' material based only on those that occur within that secondary lifecycle. This would include any processes associated with recycling the product, transport of the materials, its secondary use, and final end-of-life waste (if applicable); however, upstream impacts from the material's primary lifecycle are not counted. The cut-off approach is based on drawing clear boundaries between lifecycles in cascading systems (e.g., cases of multiple reuse or recycling phases).

In the case of compost, however, it is more difficult to determine where the boundaries should be drawn in between lifecycles. On one hand, gathering organic waste and processing it into a new product can be seen as a form of waste recycling that generates a new, valuable product. In this sense, the emissions of the composting process should be allocated to the lifecycle that uses compost as a farm input, shown by option A in Figure 5. This is consistent with the allocation method used by Venkat (2012) in their LCA of organic and conventional farming systems in California, USA.

On the other hand, compost could simply be seen as an invaluable by-product of the waste management system. It is likely that, at least for municipal composting facilities, green waste and food waste would be processed within composting systems regardless of whether a specific farm was buying this waste. The UK, for example, is legally committed to reduce the biodegradable waste sent to landfill, thus creating a need for composting as one type of alternative waste management (UK Government, 2003, 2011; Defra, 2021c). In this case, composting could simply be viewed as a waste management strategy, with its burdens attributed to the lifecycle of the primary product (e.g., to the lifecycle of the original crop production or industrial process that generated the organic waste). This scenario is depicted as Option B in Figure 5. With this approach, no environmental burdens from the composting process would be attributed to farms using this compost because it is seen simply as a waste material. The only attributed burdens would be any transport impacts from acquiring the compost from a composting facility.

Another approach that is often preferred when comparing multi-functional or co-productive processes is system expansion (ISO, 2006a, 2006b; Meier *et al.*, 2015). In this case, system boundaries are expanded so that the co-products and co-functions that arise within a new lifecycle (e.g., recycled product) are incorporated within the functional unit (Kousemaker, Jonker and Vakis, 2021). Ideally, the whole cascade, or all subsequent lifecycles of the original product, are considered. If considering just one product within the cascade, then the effects from re-processing or recycling should be distributed among all lifecycles in the cascade, often based on physical properties (Tillman *et al.*, 1994). However, as mentioned, this becomes difficult when information is only known about one product in the cascade.

Thus, a simplified method of system expansion, called the substitution method, can be used (Rehberger and Hiete, 2020). In the substitution method, the avoided impacts from the substitution of a conventional process with an alternative process are credited to the lifecycle of the alternative process or product (Kousemaker, Jonker and Vakis, 2021). System boundaries must still be clearly defined to ensure double counting does not occur. Credits should therefore be assigned to a particular lifecycle either for delivering secondary material to the next lifecycle, *or* for accepting secondary material, but not both (Rehberger and Hiete, 2020). Tillman et al. (1994) suggests that in this case, system boundaries for the particular product be set just before the recycling process up to the point where the secondary material becomes a valuable product. It should also be noted that, when employing system expansion and the substitution method, it is crucial to ensure that the systems being compared carry out equal functions; that is, that the avoided impacts being credited to a product are from a product system that performs an equal function (White, 2012).

When applying system expansion to the topic of composting, there are two main points at which the composting process, or the use of compost, substitutes a conventional process; thus, there are two main points where avoided impacts could occur (shown as Options C and D in Figure 5). The first point (Option C) follows the view that the composting process replaces a conventional waste management process (e.g., landfilling). In this case, burdens from the composting process should be allocated to the crop lifecycle using compost as an input, and then the avoided impacts of a conventional waste management process (e.g., how the organic waste would have otherwise been handled) should be subtracted from this system.

For example, in Martínez-Blanco *et al*. (2009)'s study comparing tomato cropping systems using municipal compost and conventional fertilisers, systems boundaries were expanded for the allocation of composting process emissions. Since the produced municipal compost is an output of the municipal waste management system, the composting process was considered as a replacement for conventional municipal solid waste streams (specifically, dumping of waste). The environmental burdens of the conventional waste dumping process were subtracted from the composting process, and the difference was then assigned to the tomato lifecycle. This method was selected so that only the fertilising function of different treatments (e.g., using compost in organic production vs. using mineral fertilisers in conventional production) could be compared (Martínez-Blanco *et al.*, 2009). Similarly, in Cleveland *et al.*

(2017)'s LCA on household gardens, the burdens of municipal composting of green waste and landfilling of food waste was subtracted from the emissions associated with home composting (as an alternative waste management strategy).

The second point (Option D) at which compost could be seen as a replacing a conventional process is during the use phase, where compost application as a fertility input could substitute the use of a mineral fertiliser. Following Bjarnadóttir *et al.* (2002)'s guidance for the use of LCA in the waste management sector, it is suggested that the burdens of the composting process be allocated to the crop lifecycle that utilises the compost as an input, whilst subtracting the avoided impacts from the manufacturing of conventional, mineral fertiliser. Boldrin *et al.* (2009) details how these avoided impacts could be calculated. This method has been similarly suggested to be used for allocation of manure-related emissions by Dalgaard and Halberg (2007). However, it appears to be used more within LCAs that compare waste treatment options (e.g., landfilling versus composting) (Blengini, 2008; White, 2012), rather than in LCAs comparing different farm types. This is because in agricultural LCAs comparing organic and conventional production, it is important to compare the fertilising function of different treatments (Martínez-Blanco *et al.*, 2009).

Finally, the California Air Resources Board (2017)'s method for estimating the greenhouse gas emission reductions from composting accounts for a wider range of benefits from compost. In this case, the emissions associated with avoided landfilling of biowaste *and* avoided mineral fertiliser and herbicide use are subtracted from the burdens of compost production, in addition to accounting for soil benefits from the use of compost (decreased soil erosion). However, most LCA studies usually apply only one avoided impact (i.e., landfilling or mineral fertiliser use), and not both.

As can be seen, choosing how to allocate burdens from the composting process is complicated and no true consensus has been reached by LCA practitioners. The matter becomes even more complex when considering the distinction between industrially or commercially-produced compost and compost made at home or on the farm. All farmers who used industrially-produced compost within this study paid for it, thus emphasising that there is value to this product and that it is not simply a by-product of the waste management stream. On the other hand, farmers producing homemade compost, which may have included locally-sourced manure, woodchips, or other agricultural by-products, as well as on-farm waste, did not pay for any of these inputs. This suggests that the feedstock materials are indeed waste products, in such a way that home composting could be seen simply as an alternative waste management strategy.

2.1.5.1.2 Composting allocation scenarios used in this study

Allocation procedures should be applied to a system based on scope and purpose of the LCA. Since the purpose of this LCA is to compare the different farming systems as distinct models utilising an individual set of practices, it makes little sense to regard compost as a substituted product for conventional fertiliser. Thus, the option for subtracting conventional fertiliser burdens from composting burdens (Option D, Figure 5) will not be utilised.

The remaining three allocation procedures (Options A-C, Figure 5) will be analysed separately as different scenarios, as the method of allocation could affect the reported results of this LCA. These will correspond to Scenarios 1, 2, and 3 in the reported results, respectively. It should be noted that these scenarios depict how the burdens from the actual *composting process* (i.e., the controlled decomposition) of organic material are allocated. This applies to compost produced both on and off the farm, although the material and energy inputs for these will be different. However, the emissions and burdens associated with *compost application* (i.e., the use phase) will be included for *all* scenarios.

In Scenario 1, the compost is viewed as a valuable product made from 'recycled' organic materials. Burdens from the composting process are assigned to the crop lifecycle that uses the compost as an input. If the green waste produced from this crop lifecycle is composted, then the burdens of the composting process will be applied to the subsequent crop lifecycle that uses this compost. The system boundaries for Scenario 1 are depicted in Figure 6.

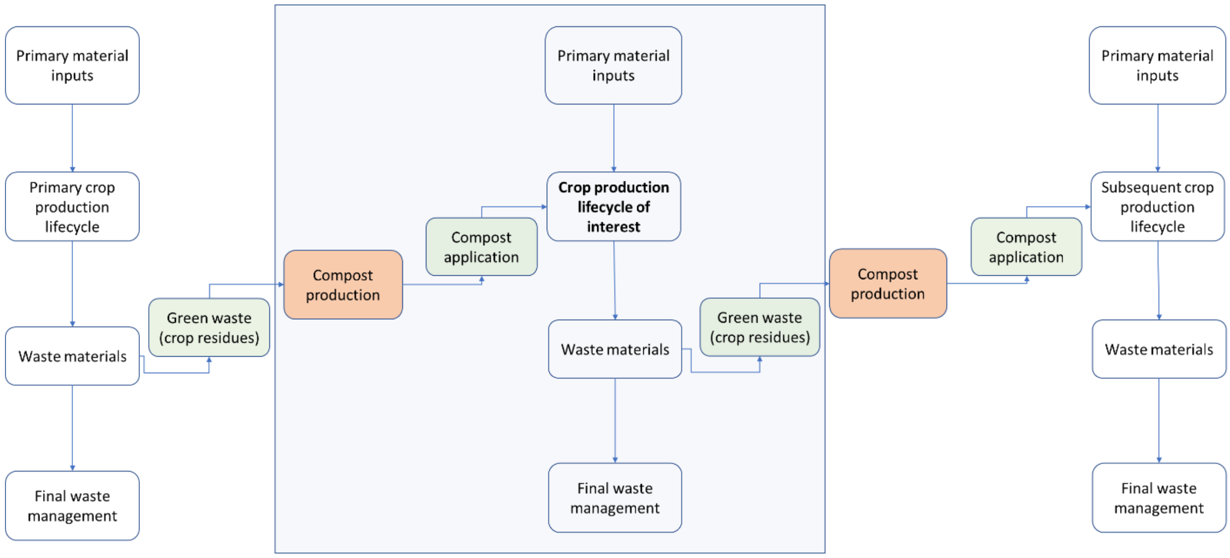

Figure 6 *– System boundaries for Scenario 1 (cut-off allocation). Burdens from the composting process are applied to crop lifecycles using compost as an input.*

In Scenario 2, compost is viewed as a by-product of the composting process, which is used as a type of waste management. Thus, burdens from the composting process are assigned to the primary crop lifecycle that produces the waste that is being composted (e.g., from the system producing compost as an output). For crop lifecycles using compost as a fertiliser input, no burdens from the production of this 'input' compost will be applied. The system boundaries for Scenario 2 are depicted in Figure 7.

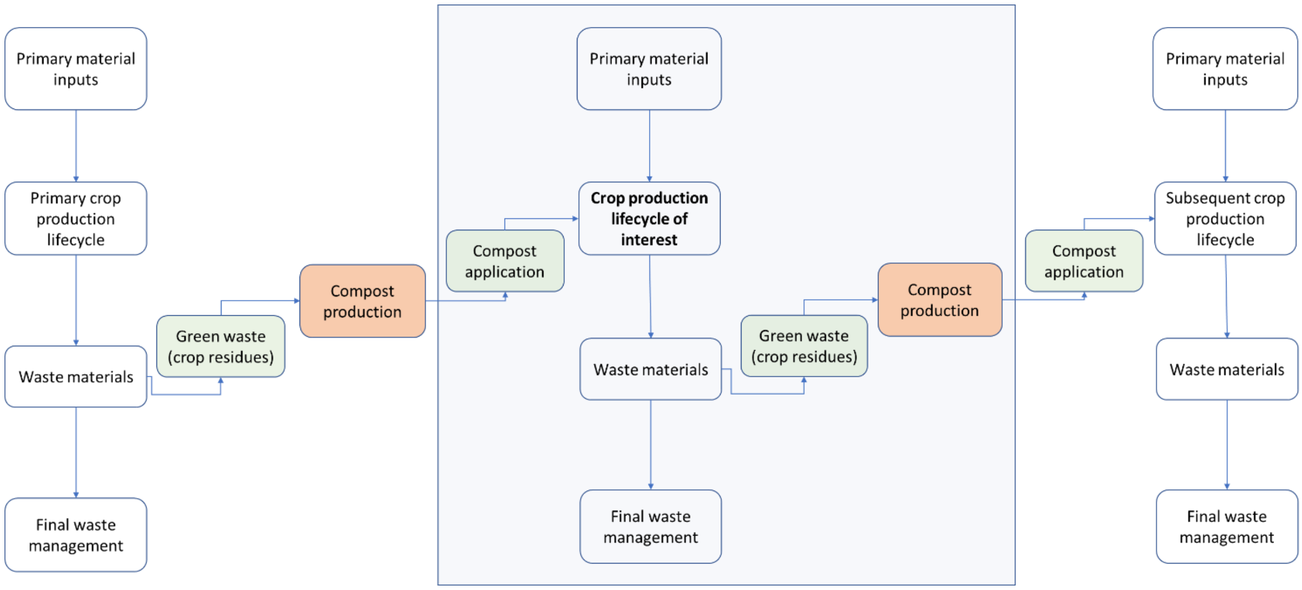

Figure 7 *- System boundaries for Scenario 2 (cut-off allocation). Burdens from the composting process are applied to the crop lifecycle that produces the organic waste being composted.*

In Scenario 3, the system is expanded to acknowledge the dual function of composting as both a waste treatment process and as a form of fertiliser production. In order to assess just the fertilising function of compost, which is what is of interest for this agricultural LCA, the avoided burdens of the alternative waste treatment option (in this case, the municipal waste stream) are subtracted from the composting process. In line with suggestions from Tillman *et al.* (1994) and Rehberger and Hiete (2020), system boundaries will be set to include the 'recycling process' for the product of interest (in this case, the compost applied on farms) and will extend to the end of the crop cultivation process, as this is the system of interest in this LCA. Consistent with Martínez-Blanco *et al.* (2009)'s LCA comparing tomato production using municipal compost and conventional fertilisers, avoided burdens will only be attributed to the compost applied on the farm as a fertiliser input, *not* to the green waste composted on farms as an output of the crop lifecycle, in order to avoid the double counting of credits. Further following this study, the burdens of composting green waste from the crop lifecycle of interest are excluded entirely by applying a cut-off at this stage, as this composting process will be considered as the 'input' recycling process for the next crop lifecycle (Martínez-Blanco *et al.*, 2009). These system boundaries for Scenario 3 are depicted in Figure 8.

The avoided impacts to be subtracted from the composting process will be based on the assumption that the organic waste used as compost feedstocks would have otherwise gone to the municipal solid waste (MSW) stream in either Georgia or England. In Georgia, landfills are the primary waste treatment for municipal solid waste and yard waste (Georgia Environmental Protection Division, 2021). Thus, the avoided impacts of landfilling organic waste will be subtracted from the composting process for the farms located in Georgia, modelled using the ecoinvent v.3.0 process for sanitary landfill waste treatment (ecoinvent Centre, 2021). For farms located in England, the avoided impacts will be calculated from the Great Britain municipal solid waste market mix as listed in ecoinvent 3.0 processes, which assumes 35% treatment via sanitary landfill and 65% treatment via incineration (ecoinvent Centre, 2021).

It should be noted that it is not necessarily the case that the green waste or food waste used to make compost would have otherwise gone to the municipal solid waste stream for all composts produced in the study. For example, for composts produced on the farm from agricultural residues and food waste, it is unlikely that the alternative scenario would be for residues to go to municipal solid waste. Alternatively, they would likely be turned into the soil or left somewhere to naturally decompose. However, if the alternative scenario was natural decomposition, then there would be no avoided burdens as it is the composting process itself that introduces additional emissions, so this option is therefore already represented (within Scenarios 1 and 2, where compost processing emissions are included). Since it is still the case that the alternative waste management option for many composts used in this study (e.g., the municipal composts) would be the municipal solid waste stream, this has been used as an option to consider potential benefits from the composting process.

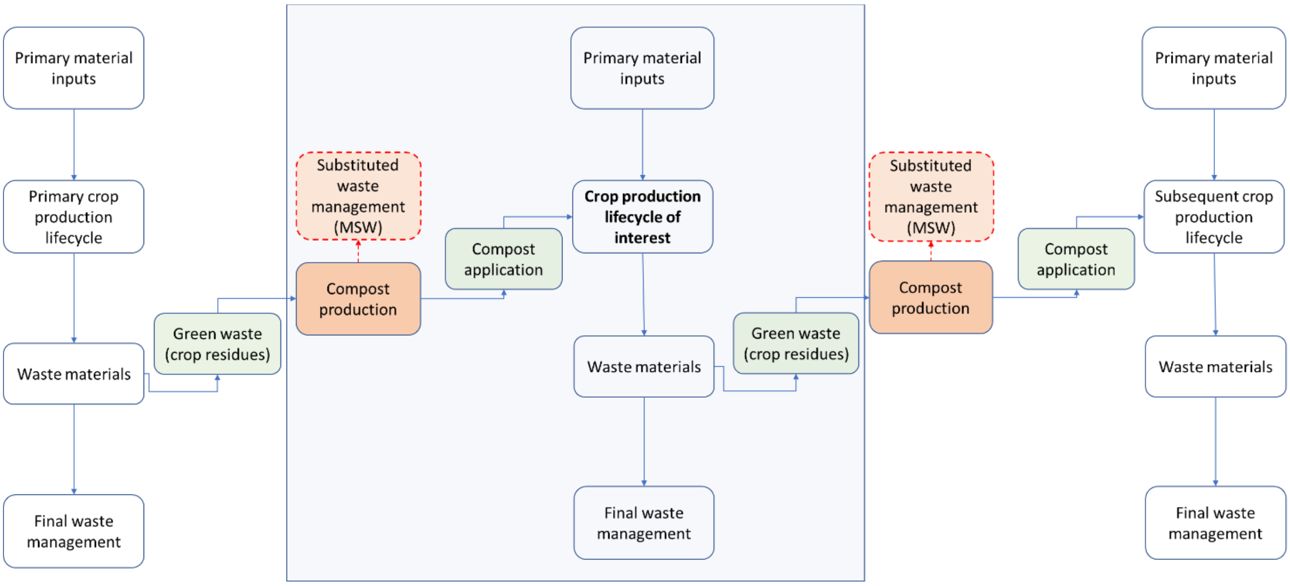

Figure 8 *- System boundaries for Scenario 3 (system expansion using the substitution method). Burdens from the composting process are applied to crop lifecycles using compost as an input, but the crop lifecycle is also credited with the avoided impacts of municipal solid waste treatment option.*

2.1.5.1.3 Manure allocation

Manure refers to faecal matter excreted by animals. Eight organic farms in this study use solid manure as a fertility input, usually in addition to other green waste composts or organic fertilisers. Most farms spread the manure fresh (perhaps keeping it in piles on the farm for a short amount of time), while three of the farms compost the manure with other green waste materials over a longer period of time (up to 24 months). The manure used is either produced from their own animals on the farm, or from nearby livestock farms or horse stables. The types of manure used by organic farmers in this study include: chicken, cow, horse, and rabbit manure.

Manure contributes to negative environmental impacts during both storage and application (Dalgaard and Halberg, 2007). If not managed properly, manure piles may contaminate underground and aboveground water bodies with nitrate and phosphate, contributing to eutrophication risk (Svanbäck *et al.*, 2019; Kamilaris and Prenafeta-Boldú, 2021). Manure can also release nitrous oxide and ammonia emissions to the atmosphere (Davidson, 2009;

UK Environment Agency, 2011). Ammonia emissions occur when the manure comes into contact with air, either while it is being stored in piles or while it is spread on the field (Defra, 2018).

As employed throughout this LCA, the cut-off allocation method is used to allocate the burdens from the production of manure from animals. Manure is a waste product, produced as a by-product of an animal's lifecycle. It has little value, usually acquired by farmers for free, and in fact its over-production on livestock farms is often an issue. Thus, none of the emissions associated with its production (i.e., the animal lifecycle) should be attributed to the crop lifecycle.

The burdens that will be applied to all manure used for a crop lifecycle include: transport impacts associated with the delivery of manure from nearby farms; emissions from manure application as a fertility input on the farm; and material, energy, and fuel use associated spreading the manure (e.g., tractor use). It should be noted that application emissions are included as this is directly related to the use of manure as a fertiliser on the farm, and without the manure, the farm would have had to use another organic fertiliser that would have likely generated similar emissions from application. Manure application emissions are calculated the same as for all organic inputs, detailed in Section 2.1.9.

However, there are also emissions that occur when manure is collected and piled. The allocation of emissions from the *storage* of manure is handled differently based on whether the manure is stored temporarily in piles and then spread fresh, or if it is composted over time with other green waste materials as a component of a farm's homemade compost.

In the first case, gaseous emissions generated from temporary, short-term storage of fresh manure in piles, before it is incorporated (fresh) into fields, will not be included in any scenarios. It is assumed that these emissions should be attributed to the animal lifecycle, and thus should not be allocated to the crop lifecycle using manure as a fertility input, as suggested by Dalgaard and Halberg (2007) and used by Liu et al. (2010) in an LCA on organic pear farms using manure inputs. Most farms in this study that use manure incorporate it fresh and do not compost it, so it should be assumed that it is only stored in a pile on the vegetable farm for a short period of time (generating negligible emissions), before being applied to the soil, where application emissions are then counted.

Even if emissions do occur while the manure is stored in piles on a vegetable farm, it can be assumed that these emissions would have happened regardless of whether the manure was used by the vegetable farm. Indeed, it is standard practice on livestock farms to pile manure and then spread it on fields as a fertility input and soil builder (UK Environment Agency, 2011; Svanbäck *et al.*, 2019); thus, it could be assumed that if the vegetable farm had not taken the manure, the same (or potentially even worse) emissions would have occurred on the livestock farm. The reason that vegetable farms are able to access manure as a fertility input at all is because many livestock farms produce such large amounts of animal manure that they cannot spread it all across their own fields, and thus must export their manure (Svanbäck *et al.*, 2019; Kamilaris and Prenafeta-Boldú, 2021). This presents the view that manure is an environmental problem generated by the livestock industry, and thus the burdens of manure should be attributed to livestock lifecycles.

For composted manure, where manure is added to other green waste or food waste produced on the farm to make a compost, then the allocation of emissions will follow that as discussed in Section 2.1.5.1.2 for biowaste composts. Emissions from the whole compost pile (including the weight of biowaste and manures) will be allocated in three different scenarios, and in this way, it will be seen how attributing or not attributing burdens of the composting process affect final results. In the scenarios where composting burdens are attributed to the crop lifecycle, then the gaseous emissions that result from the biowaste composting process will be applied to the total amount of compost used, as if biowaste were the sole input. No additional consideration will be made for particular or extra emissions from the use of manure in the compost instead of biowaste. Although in some models, composted manure is listed to have higher levels of emissions for certain gases (ADEME, 2020), the "additional" burden of using the manure in the compost will not be attributed to the vegetable farms. This is because the vegetable farms are providing a solution for the over-production of manure on livestock farms, so any additional negative burdens should be attributed to the animal lifecycle that is creating this waste problem, not to the vegetable farmer providing the solution. However, composting emissions for the whole compost amount used should still be counted, because if the vegetable farmer did not have manure as a compost feedstock, then they would theoretically have had to replace the manure with other biowaste material to generate the amount of compost needed for their farm.

Of course, it could be argued that all storage / composting emissions from manure should attributed to the animal lifecycle. In Scenario 1, where compost input burdens are attributed to the crop lifecycle, this will be assessed in a sensitivity analysis where manure components of compost are excluded completely. Further, in Scenario 2, where compost input burdens are not allocated to the crop lifecycle, this will be applied again.

In the system expansion scenario (Scenario 3), the avoided burdens of organic waste being sent to the municipal solid waste stream, instead of being composted, are applied to compost inputs. These avoided burdens will *not* be applied to the manure portion of the compost. Instead, the gaseous emissions from composting manure will just be excluded. This is because the alternative scenario would be the composting or piling taking place on the livestock farm, which would theoretically produce the same emissions. Thus, the avoided burdens of this would cancel out the burdens of composting manure on the farm.

In summary, no emissions from the piling of manure on vegetable farms that use fresh manure as a fertility input will be allocated. For farms that compost the manure with other green wastes over the long-term, composting emissions will be allocated within the compost allocation scenarios (defined in Section 2.1.5.1.2), with sensitivity analyses taking place within these scenarios to see the impact of allocating or not allocating manure composting emissions. The emissions from the application of manure to soil for the crop lifecycle will be counted in all cases.

2.1.5.2 Combined heat and power (CHP) allocation

A combined heat and power (CHP) unit produces heat, electricity, and carbon dioxide from the burning of natural gas. When CHP is used for horticultural glasshouse or greenhouse production, generally all the heat and CO_2 is used within the glasshouse, and electricity is often produced in excess (Blonk *et al.*, 2010). This surplus electricity is usually sold back to the national grid. Thus, the main purpose of a CHP unit for glasshouse production is to provide heating as this is the main energy requirement (Almeida *et al.*, 2014; Marttila *et al.*, 2021). This creates an issue with how to allocate the burdens from the CHP unit between heat and electricity, and how to account for the surplus electricity produced. It should be noted that usually CO_2 is disregarded in this allocation since the main point of a CHP unit is to produce electricity and heat, although the CO_2 is used within the glasshouse and thus allows growers to avoid the need to buy in liquid CO_2, a standard practice in glasshouse horticulture.

Several methods of allocating burdens between heat and electricity in CHP have been used in LCA studies and databases. These include mainly energy, exergy, and economic allocation, as well as using system expansion methods (Dones *et al.*, 2007). In some cases, all burdens may be allocated to heat or electricity, if one is viewed as the main product, although this is not common (Dones *et al.*, 2007; Laurent, Espinosa and Hauschild, 2017). Mostly, exergy, energy, or system expansion scenarios are used (Laurent, Espinosa and Hauschild, 2017).

Ecoinvent v.3.7.1 processes in cut-off allocation scenarios use exergy allocation, as a way to assess energy quality, in which burdens are assigned based on the exergy content of the heat and electricity (Dones *et al.*, 2007; ecoinvent Centre, 2021). The benefit of this method is that it allocates based on the extent for each form of energy to be converted to work (Laurent, Espinosa and Hauschild, 2017). This method, however, allocates higher burdens to electricity instead of heat (contrary to energy allocation), as heat has a lower exergy content (Dones *et al.*, 2007; Primas, 2007). Since heat is actually the main energy type required by heated glasshouse production, this allocation method is not ideal for this case. Thus, energy allocation, which alternatively applies the majority of the burdens to heat production (Laurent, Espinosa and Hauschild, 2017), is more optimal for the case of heated glasshouses.

Two methods of CHP allocation are thus explored in this LCA. These include allocation based on energy content and the use of system expansion, in which surplus electricity is viewed as an avoided product (Blonk *et al.*, 2010; Vermeulen and Van Der Lans, 2011; Antón *et al.*, 2012). Energy allocation for CHP will be used in the cut-off allocation Scenarios 1 and 2. The system expansion method will be used within the general system expansion Scenario 3.

Energy allocation is dependent on the electrical and thermal efficiencies of the CHP unit and the heating efficiency of the greenhouse (Blonk *et al.*, 2010). First, the input of natural gas is allocated to electricity and heat based on these efficiencies. Then, the amount of natural gas allocated to electricity production, to produce the *surplus* electricity, should be subtracted from the total amount needed.

In this LCA, it is assumed that the natural gas input to the CHP unit completely and exactly satisfies the heating demand, but produces surplus electricity. In this case, for every kWh of surplus electricity produced by the CHP unit, the relevant amount of natural gas allocated to the electricity component should be subtracted from the gas input. This method is less commonly used within agricultural LCAs, but has been explored in some studies (Blonk *et al.*, 2010; Vermeulen and Van Der Lans, 2011; Antón *et al.*, 2012).

In the system expansion method, it is assumed that the supply of a co-product replaces a conventional process elsewhere, resulting in avoided impacts (Blonk *et al.*, 2010). In the case of CHP, the surplus electricity that is produced and sold back to the national grid results in avoided burdens when using system expansion. The avoided burden is the production of the same amount of surplus electricity through the national grid, which should be subtracted from the burdens of the CHP system (BSI, 2011). This is method is commonly employed among agricultural LCAs investigating glasshouse production with CHP (Williams, Audsley and Sandars, 2006; Blonk *et al.*, 2010; Vermeulen and Van Der Lans, 2011; Antón *et al.*, 2012; Röös and Karlsson, 2013; Almeida *et al.*, 2014; Marttila *et al.*, 2021), and is the method suggested by the British Standards Institution when assessing emissions from energy production utilising CHP (BSI, 2011).

2.1.6 Summary of sensitivity analyses

Further to these three scenarios, certain sensitivity analyses will be explored based on areas where there is uncertainty and inconsistency around modelled processes in LCA literature. In particular, three main areas are investigated, which include: additional considerations for composting burdens, different nitrate leaching models, and assumptions used for soilless system emissions, the latter of which only applies to UK conventional tomato production. Due to the wide range of sensitivity analyses to investigate, these will just be applied to Scenario 1 as the default scenario.

In further detail, the specific items tested within the sensitivity analysis include:

- Composting
    - Exclusion of all fugitive emissions from the compost pile.
    - Inclusion of average carbon sequestration factor for compost, as avoided CO_2; the value used is 0.101 kg CO_2-eq (kg compost wet weight)$^{-1}$, as estimated through the consideration of several literature sources by (Boldrin *et al.*, 2009).
    - Inclusion of a maximum carbon sequestration factor for compost, which estimates additional carbon savings from compost application, such as increased soil water retention and reduced erosion; the value used is 0.675 kg CO_2-eq (kg compost)$^{-1}$ as per (Saer *et al.*, 2013).
- Nitrate emission models
    - Testing four additional nitrate leaching model in comparison to the default model used in this LCA, which is the de Willigen 2000 model (de Willigen, 2000). The other four models tested include: the SQCB-NO_3 model (Emmenegger, Reinhard and Zah, 2009); the Smaling 1993 model (Smaling, Stoorvogel and Windmeijer, 1993); the Poore-Nemecek model (Poore and Nemecek, 2018b); and the IPCC 2019 Tier 1 emission factor for nitrate leaching (IPCC, 2019). For more detail on these models, see Section 2.1.9.5.2.
    - Modelling leaching of fertilisers using only the soluble N fraction, instead of the total N fraction; this is tested only within the default nitrate leaching model (de Willigen, 2000).
- Soilless systems
    - Testing different values for direct N_2O emissions during crop cultivation. The default value used is 0.0087 kg N_2O-N (kg applied N)$^{-1}$, based on a literature average (Daum and Schenk, 1996a; Yoshihara *et al.*, 2014; Llorach-Massana *et al.*, 2017; Karlowsky *et al.*, 2021). Minimum and maximum values as measured in literature are also tested, respectively 0.001 kg N_2O-N (kg applied

N)$^{-1}$ (Karlowsky *et al.*, 2021) and 0.046 kg N_2O-N (kg applied N)$^{-1}$ (Yoshihara *et al.*, 2014).

- Exclusion of all N-related emissions to air during crop cultivation (N_2O, NH_3, and NO_2), due to the uncertainty around whether these emissions occur in soilless systems (Almeida *et al.*, 2014; Dias *et al.*, 2017).
- Inclusion of nitrate leaching and associated N_2O emissions from closed hydroponic systems. In the default method, leaching is assumed to only occur from open hydroponic systems.
- Testing the proportion of open and closed hydroponic systems. The default assumes 50% open and 50% closed systems; the sensitivity tests the assumption of 100% open and 100% closed. This influences estimated nitrate leaching amounts, water use, and fertiliser use.

The assumptions and uncertainties that exist for these areas, as seen in LCA literature, are further discussed within their relevant lifecycle inventory sections.

2.1.7 Lifecycle inventories (LCI)

This section provides an overview of the lifecycle inventories (LCIs) used to model relevant resource and material inputs, as well as waste and emission outputs, from kale and tomato lifecycles. Section 2.1.7.1 provides a detailed overview of all case study farms included within this LCA project. Section 2.1.7.2 then gives a basic overview of the main flows modelled within this LCA, how these are allocated to specific crops, any assumptions made, and the main data sources used. This section additionally identifies any items specifically excluded from this LCA.

Section 2.1.7.3 and Section 2.1.7.4 then provide the major flows modelled for kale and tomato lifecycles, with amounts designated mainly using primary data from farmers. The LCIs for kale and tomato crops provided within these sections are then divided into four main phases of production, as outlined within the system boundaries: seedling production; crop cultivation; processing and storage; and distribution to the final point of sale. LCIs for seedling production are provided per seedling transplanted; for cultivation, per gross cultivation area (m^2); and for processing and transport, per kg sellable crop. These have been selected so that information from these LCIs can be easily compared or used in other relevant studies, based on yields or cultivation areas on other farms. The tomato LCIs include additional information for how the fuel use for combined heat and power and the production of surplus electricity is calculated and allocated for UK conventional tomato production. Additional LCIs are then provided in subsequent sections for specific infrastructure (2.1.7.5) and compost (2.1.7.6), the latter of which specifies the composting burdens that are allocated differently depending on the three allocation scenarios employed within this LCA.

2.1.7.1 Farm case studies

An overview of the main characteristics of the farm case studies are included in Table 1 for U.S. farms and Table 2 for UK farms. These tables provide information on regional geographical location of the farms, but specific locations of farms are excluded maintain confidentiality. In Georgia, regions are listed based on the twelve geographical regions of Georgia (https://www.georgia.org/regions). In England, this is based on the UK Met Office's climate districts map (https://www.metoffice.gov.uk/research/climate/maps-and-data/about/districts-map). North, south, east, and west are depicted as N, S, E, and W, respectively.

Information on farm area, in acres, is also provided, which includes total farmland, as well as total cropped area. The cropped acreage includes the total land area used for crop cultivation, which includes the crops studied in this LCA (kale and tomatoes) as well as any other crops that the farm may produce. Other land area on the farm may include uncultivated fields, undisturbed woodland area, or managed woodland areas. It is then specified whether the farm listed is evaluated within the kale or tomato LCA, or both. While all organic farms grew both crops, in some cases a crop lifecycle may not have been evaluated if it did not constitute an average year for the farm, for example, if there was a crop failure that year (e.g., US-PU-O4 for tomatoes and UK-U-O-1 for kale). However, it should be noted that all farms listed (including the conventional farms) grow a wider range of vegetables than just kale or tomatoes, and in some cases also grow row crops (such as peanuts or cotton in Georgia) or arable crops.

The management type listed distinguishes whether the farm is organic or conventional, which is also distinguished within the farm identification label, as 'O' or 'C'. For organic farms, it is further specified whether or not the farm holds an organic certification, or whether it has just self-identified as following organic practices. It can generally be seen that the smaller-scale urban and peri-urban farms tend not to hold an organic certification, which is usually due to issues with cost. In the U.S., 'certified naturally grown' is another certification thatfollows similar guidelines as organic certification, but employs a peer-to-peer certification process to reduce costs (Certified Naturally Grown, 2023). For conventional farms, it is also specified whether production occurs in the field or in glasshouses, the latter of which only applies to UK conventional tomato production. It should also be noted that, for UK conventional tomato production, only one case study site is considered, but using two cases of energy sources. This is designated as R-C-NG, when using natural gas heating and the national electricity grid, and R-C-CHP, when using combined heat and power.

Finally, the main markets of sale used by the farm for tomato and kale production are listed, although this may differ slightly between the specific crops in question on some organic farms. Common markets of sale for organic farms in Georgia include: farmers' markets, community-supported agriculture (CSA), on-farm shops or stands, and direct sale to restaurants. CSA is a type of agricultural model where individuals purchase of share of the farm's harvest in advance, usually paying a set fee at the beginning of the growing season, and in return receive a share of the farm's harvest on a regular basis (e.g., weekly or bi-weekly). In some cases, farmers deliver these shares to a central location (as in the case of US-R-O-2), or they may be distributed from the farm (as in the case of US-PU-O-1). As for consumers travelling to shops, no transport burdens are applied for individuals picking up the produce on the farm. Finally, some of the organic farms in Georgia also sell a small portion of produce through an organic wholesale market in Atlanta, which then distributes to restaurants in the city. Peri-urban farms in Georgia, located in the Atlanta metropolitan region, tend to sell their produce at nearby markets rather than in the heart of the city. In contrast, the rural organic farms often sell a portion of produce in nearby towns as well as in Atlanta. The conventional farms either sell through a wholesaler that then distributes produce to supermarkets around the region, or may sell to supermarket distributors directly.

UK organic farms employ similar markets of sale to U.S. organic, although selling through vegetable ('veg') boxes is more common. This model is similar to a CSA, in that customers routinely receive a box or bag of produce from the farm, but the main difference is that consumers pay for each box rather than paying a lump sum for a share of the harvest before the growing season. Like CSA, veg boxes may be delivered directly to consumers (home delivery) or delivered to pick-up points within the town or city. Another difference for UK rural organic farms is that they mainly sell produce in their nearest local town or city, rather than selling through the central capital city of London; this is in contrast to U.S. rural organic farms, which all sell a portion of their produce in the capital of Georgia, as well as in their local towns. The UK conventional farms mainly sell produce directly to supermarket distributors, rather than selling through wholesalers. However, some farms do sell to wholesale distributors or food processors, that will then sell the produce on to supermarkets (mainly) or restaurants. UK-R-C-1 also sells through online wholesalers and food box delivery providers. For all UK conventional farms, produce is distributed throughout Great Britain (i.e., England, Wales, and Scotland), although most is generally sold in England.

Table 1 *– Characteristics of U.S. farms included in the LCA study*

U.S. Farms	Geographical region	Total farm acreage	Total cultivated crop acreage	Crops evaluated	Management type	Markets of Sale	Location of sale
U-O-1	Atlanta	4	1	Kale & tomatoes	Organic practices	Farm stand, CSA, restaurants	Atlanta
PU-O-1	Metropolitan Atlanta	4	0.1	Kale & tomatoes	Organic practices	CSA	On farm
PU-O-2	Metropolitan Atlanta	n/a [a]	0.50	Kale & tomatoes	Certified naturally grown	Farmers' market, CSA (on another farm)	Surrounding towns / suburbs
PU-O-3	Metropolitan Atlanta	1	1	Kale & tomatoes	Mostly organic practices	Farm shop	On farm
PU-O-4	Metropolitan Atlanta	n/a [a]	0.20	Kale	Organic practices	Farmers' market	Surrounding town / suburbs
R-O-1	Middle Georgia	53	2.75	Kale & tomatoes	Certified organic	Farmers' markets, restaurants	Local towns, Atlanta
R-O-2	East central Georgia	175	7	Kale & tomatoes	Certified organic	Farmers' markets, CSA, wholesale	Local town, Atlanta
R-O-3	Northwest Georgia	16	8	Kale & tomatoes	Certified organic	Farmers' markets, restaurants, wholesale	Local town, Atlanta (for wholesale)
R-C-1	Southwest Georgia	1,450	1,200	Tomatoes	Conventional (field)	Wholesale, distributed to supermarkets	Wholesale in Atlanta, distributed in southeast U.S.
R-C-2	Northeast Georgia	600	500	Tomatoes	Conventional (field)	Wholesale, distributed to supermarkets	Wholesale in Atlanta, distributed across East coast
R-C-3	Southeast Georgia	Not reported	6,500	Kale	Conventional (field)	Direct to supermarket distributors	Across East coast

[a] Note that total farm acreage is not listed for these farms as they both grow produce mainly on neighbours' front / backyards; thus, the cultivated area is the total area.

Table 2 – Characteristics of UK farms included in the LCA study

UK Farms	Geographical region	Total farm acreage	Total cultivated crop acreage	Crops evaluated	Management type	Markets of Sale	Location of sale
U-O-1	SW England	1.2	1.2	Tomatoes	Organic practices	Restaurants, shops	Local city
U-O-2	SW England	12	3	Kale & tomatoes	Organic practices	CSA (home delivery)	Local city
PU-O-1	E & NE England	5	1.2	Kale & tomatoes	Certified organic	Shop, veg box (home delivery)	Local city
PU-O-2	E & NE England	10	2.5	Kale & tomatoes	Certified organic	Restaurants, veg box (home delivery)	Local city
PU-O-3	E & NE England	15	1	Kale & tomatoes	Organic & permaculture practices	Veg box (pickup points)	Local city
PU-O-4	E & NE England	10	9	Kale & tomatoes	Certified organic	Farmers' market, veg box (pickup points)	Local city
R-O-1	NW England	5	4	Kale & tomatoes	Certified organic	Shops, wholesale market	Nearest city
R-O-2	SE & central S England	31	27	Kale & tomatoes	Certified organic	Farmers' market, farm stand	On farm, nearby towns, nearest city
R-C-1	Midlands	250	50	Kale	Conventional (field)	Supermarkets, food service, recipe boxes	Great Britain
R-C-2	E & NE England	150	140	Kale	Conventional (field)	Wholesale, food processors, supermarkets	Great Britain
R-C-3	NW England	400	170	Kale	Conventional (field)	Supermarkets	Great Britain
R-C-4	SW England	Not reported	6,500	Kale	Conventional (field)	Supermarkets	Great Britain
R-C-5	East Anglia	Not reported	2,080	Kale	Conventional (field)	Supermarkets	Great Britain
R-C-NG / R-C-CHP	SE & central S England	68	65	Tomatoes	Conventional (glasshouse, soilless)	Supermarkets	Great Britain

2.1.7.2 Overview of main processes, assumptions, and exclusions

This section provides an overview of the main processes and flows modelled within kale and tomato crop lifecycles, the data source used, and any assumptions made. The main data sources used for LCIs has been previously discussed in Section 2.1.2.1. Foreground data on the amounts of inputs used is provided by the case study farms. Background data, such as that modelling the burdens and emissions associated with material, fertiliser, and pesticide production, infrastructure, transport, and waste treatment is modelled using secondary databases. These include: ecoinvent v.3.7.1, (ecoinvent Centre, 2021), AGRIBALYSE v.3.0 (ADEME, 2020), Agri-footprint v.4.0 (mass allocation) (Blonk Sustainability, 2017), and the World Food LCA Database (WFLDB) v.3.5 (Quantis and Agroscope, 2019). Table 3 provides an overview of the main database used for each major process modelled.

Table 3 *– Secondary LCI databases used for major processes modelled*

Flow	Data source
Materials and material processing	ecoinvent v.3.7.1
Electricity use	ecoinvent v.3.7.1
Fuel use (petrol, diesel, propane, natural gas) and combustion emissions	ecoinvent v.3.7.1
Tap water	ecoinvent v.3.7.1
Glasshouse infrastructure	AGRIBALYSE v.3.0
Shed infrastructure	AGRIBALYSE v.3.0 / ecoinvent v.3.7.1
Packhouse / distribution centre infrastructure	ecoinvent v.3.7.1
Refrigerators	WFLDB v.3.5
Organic fertilisers	AGRIBALYSE v.3.0
Mineral / synthetic fertilisers	ecoinvent v.3.7.1
Pesticides	Agrifootprint v.4.0 / ecoinvent v.3.7.1
Two-wheel tractors	AGRIBALYSE v.3.0
Four-wheel tractors and implements	ecoinvent v.3.7.1
Transport	ecoinvent v.3.7.1
Waste treatment	ecoinvent v.3.7.1

The main processes modelled within each phase of production has been previously highlighted in Figure 2. The following sections provide information on the major assumptions used when modelling these flows. Information about the modelling of agricultural emissions (e.g., from fertilisers, pesticides, and crop residues) is provided separately in Section 2.1.9. Waste treatment is considered for all items disposed of on the farm site or farm packhouses. Waste occurring at distribution centres, points of sale, and after consumption (e.g., in households) is not considered due to lack of information in the first case and being outside the scope of the LCA for the latter two cases.

2.1.7.2.1 Crop waste

Harvest waste from crops is modelled based on estimations provided by farmers. Harvest waste is considered as produced crop that may have been sold, if not for other circumstances that rendered the crop unsellable. It is listed within LCIs as a percentage of all produced crop (by weight). In most cases, harvest waste is not explicitly measured by farmers, so there is a degree of uncertainty surrounding these measurements. Processing waste is considered to be that which is graded out within packhouses during the packaging process. It is listed within LCIs as a percentage of all harvested crop (by weight). Processing waste is explicitly measured on some farms, whilst other farms may provide an estimate for processing waste. Amounts of crop waste designated as harvesting waste and processing waste vary depending on how grading or sorting is performed on the farm. For example, some farmers choose to harvest all crop and then grade it within the packhouse, whilst other farmers prefer to do most of the grading during harvest; thus, in some cases harvest waste may be relatively low while processing waste is relatively high, and vice versa.

Crop waste is handled differently across farms. Harvest waste is almost always returned back to the soil, and in some cases processing waste is also incorporated back into the field soil. When this happens, crop waste is considered as an N input to the soil within the crop lifecycle. Section 2.1.9.2.2.2 details how N is calculated for harvest waste and other crop waste so that agricultural emissions from this N input can be applied. In other cases, harvest and processing waste may be composted, and thus emissions from composting (detailed in Section 2.1.7.6.4) will be applied during allocation Scenario 2. Crop waste may also be fed to animals on the farm or on nearby farms; given to employees or eaten on the farm; or donated to churches or charities. For these cases, no burdens or emissions are considered, other than transport burdens if relevant. Only in one case (UK conventional tomato production) is processing waste landfilled. It should also be noted that, in some cases, tomatoes may be graded as 'seconds' and sold either at reduced prices at market (for organic farms) or to food processors (in the case of the UK conventional farm); as these tomatoes may still be sold, they are not considered as waste in this LCI.

Waste during transport is assumed to be zero within this LCA due to limited information on waste amounts that may occur for longer supply chains (e.g., to supermarkets). Finally, any unsold crop waste is also not considered within this LCA, as this is considered outside the scope.

2.1.7.2.2 Materials, fertilisers, and pesticides

Amounts and types of materials, fertilisers, and pesticide active ingredients used are modelled based on specific product information provided by farmers. For all of these items, the transport during production (e.g., to manufacturing centres and from manufacturing centres to points of sale) is estimated within relevant ecoinvent market processes. Transport of items from the shop to the farm (where relevant) is modelled based on specific information provided by farmers about the supplier they use and the method of transport used for delivery to the farm. Packaging for fertilisers and pesticides is based on ecoinvent and Agrifootprint v.4.0 processes, respectively. End of life waste is considered for all relevant materials. This is assumed based on waste treatment designated by the farmer, or if this information was not provided, modelled based on similar farms in the same country. However, end of life waste is not considered for the packaging of the crop, as it is assumed that this will be disposed of by consumers, and thus is outside the scope of this LCA.

Materials are allocated to the crop of interest based on time of use for the crop and material lifetime. Material lifetimes are based primarily on information provided by farmers or suppliers; when this was not possible, then lifetimes were estimated based on information from other growers, secondary databases, or other literatures sources. An additional consideration is that for reclaimed materials, as where farmers use materials that have been donated or reclaimed from other farms or businesses. These materials are still listed within the LCIs, although designated specifically. Reclaimed materials are considered as recycled materials, and thus are not allocated any production burdens; however, any relevant transport and end of life burdens are included. However, if the item was acquired whilst at a location for another purpose (e.g., while selling their produce at market), then transport burdens are not considered. Additionally, end of life waste is not considered if the item is to be disposed of outside system boundaries (i.e., by the consumer).

All mineral fertiliser inputs are modelled using ecoinvent processes. However, pesticides are modelled using two databases so that they could be modelled as products, not solely as active ingredients. Information about the active ingredients, solvents, and inert ingredients (co-formulants) within each pesticide product was derived from material safety data sheets from suppliers. Two databases were used to build pesticide models. Agrifootprint v.4.0 processes model pesticides as products, and thus these processes were used as a ‘shell’ to provide information about inert ingredients and solvents when these were unknown for a specific product. When possible, ecoinvent processes were used to provide detailed data about the production of specific active ingredients, since this is not considered within the Agrifootprint processes. In these cases, the energy use for pesticide product mixing and blending is applied separately, which is assumed as 20 MJ (kg active ingredient)$^{-1}$ based on Green (1987).

2.1.7.2.3 Cover crops

The main flows considered for cover crops are land use, seed inputs (modelled using ecoinvent processes with amounts provided by farmers), and any tractor operations, such as mowing or seeding the cover crop. Cover crop input flows are allocated to the crop of interest based on their utility for other cash crops within the crop rotation; for example, in cases where the farmer specified that the cover crop is grown mainly as an input for kale, then all associated flows are allocated to the kale crop.

2.1.7.2.4 Water

Water use is considered mainly for irrigation purposes, the spraying of pesticides or other plant products, and washing the crop. Water may be pumped from groundwater or surface water sources (e.g., rivers and ponds), for which energy use is also estimated. Some smaller farms in this study also utilise tap water; this is modelled using ecoinvent v.3.7.1 processes, which considers relevant energy use and infrastructure. Thus, listed energy use amounts for cultivation and processing as provided within the farm LCI tables will not include the energy use associated with tap water, as this is modelled within background ecoinvent processes.

Additionally, some organic farms harvest rainwater to use as a water source for crop cultivation. In this case, water use amounts have been provided within LCIs (and designated as rainwater), but it should be noted that rainwater is *not* considered as contributing to the water consumption impact category within the ReCiPe LCIA method. Rainwater falling directly onto the field is also not modelled and not considered as a water consumption flow.

2.1.7.2.5 Energy use

Electricity use across farms is modelled using relevant ecoinvent processes based on geography. Specifically, the SERC electricity grid (southeast U.S.) is used for Georgia farms and the GB (Great Britain) grid for English farms. Electricity use amounts are based on information provided by farmers, although this was often provided in one of two ways, which creates uncertainty. Some farms provided information on total energy use for the farm or the packhouse operations, which was then allocated to the crop of interest using mass allocation (proportion of crop out of total harvested crop on the farm over the year by mass), unless another estimation was specifically provided by the farmer. In other cases, farmers provided details about specific sources of energy use (e.g., from pumps, specific machinery, fans, refrigerators, etc.); equipment wattage ratings and times of use were then used to approximate electricity use (kWh). The different ways that energy use were estimated likely provides an overestimation for the prior case, where total energy uses across the packhouse or farm were provided, and an underestimation for the latter case, where specific times of use were assumed.

2.1.7.2.6 Fuel use

Fuel use during crop cultivation includes all on-farm tractor and transport operations, including any contract work. Some pumping systems also rely on fuel rather than electricity. Vehicle transport is considered only for transport between fields or growing sites. Transport of employees from their homes to their job sites is not considered within the scope of this LCA.

Diesel is main fuel source used across farms for all on-farm cultivation operations, although petrol is also used as a fuel source in a few cases, mainly for U.S. vehicles (pick-up trucks) and some two-wheel tractors and hand mowers. Propane is also used for flame weeding on some U.S. organic farms and for forklifts during packhouse operations for UK conventional farms.

Fuel use amounts may be provided by farmers based on figures for the whole farm over the year. In this case, fuel amounts will be allocated to the crop of interest based usually on mass allocation (weight of crop out of all crops produced), unless farmers have provided an alternative estimate. This is generally the case for propane use in packhouses, and for diesel use across some farms. However, in most cases fuel use is estimated based on specific operations performed. For example, diesel use from tractor operations has been calculated using equations provided by Grisso *et al*. (2014), which are based on the horsepower (HP) rating of the tractor and the amount of time (in hours) the tractor is used. Times of use for each tractor have been provided by farmers. Estimates of fuel use from on-farm vehicle transport are either provided by the farmer; estimated based on mpg ratings of the vehicle used and transport distances; or estimated from models in ecoinvent, for the specific vehicle type used.

2.1.7.2.7 Capital goods, machinery, and excluded items

For this LCA, the modelling of capital goods or equipment follows the guidance of Blonk *et al.* (2010) for assessing the carbon footprints of horticultural products, which suggests that only underlying processes in the supply chain (such as production of capital goods) for which sufficient information is available should be included within the model. Burdens from the *use* of equipment is always included (e.g., energy uses), but not all equipment used on farms had sufficient information to be modelled.

The main equipment types modelled in this LCA include: tractors and implements; vehicles, as included within ecoinvent transport processes; rainwater harvesting equipment (e.g., pipes and tanks); and refrigerators and air conditioning units used for cool rooms (see: Section 2.1.7.5.2). However, certain items have been excluded due to lack of appropriate information and models regarding their material make-up, production, and lifetimes. Pumps have not been modelled as part of the irrigation equipment, although pump energy use and other irrigation equipment (e.g., piping) have been considered. Hand tools that may be used on small farms (e.g., scythes, watering cans, hand ploughs) have not been modelled, as an initial sensitivity check rendered the contributions from these items as negligible. Additionally, large equipment used in nurseries (such as seeding machines) were not included due to limited information. Similarly, capital goods within packhouses, such as chopping machines, conveyor belts, and washing facilities have not been modelled.

Cultivation equipment weights (tractors and implements) are modelled based on the specific brands used by farmers, with information provided by suppliers. These are allocated to the crop lifecycle based on the time each piece of equipment is used for the crop, out of its total lifetime in hours. In cases where farmers could only estimate an equipment lifetime in years, then yearly allocation is based on the area the crop takes up across the total farm cropped area. In cases where farmers could not provide an estimated lifetime, this has been estimated as 3,000 hours for two-wheel tractors (Ademiluyi *et al.*, 2007) and 9,000 hours for all other tractors, based on information provided by farmers, as well as estimates within ecoinvent and AGRIBALYSE databases. The lifetime of implements, when not provided by farmers, is estimated as 800 hours for tillage machinery, based on ecoinvent processes; for other equipment, lifetimes have been estimated based on Redman (2019). All cultivation machinery has been modelled using ecoinvent v.3.0 processes. Tractors and implements are considered to be mostly steel, although for the prior, rubber use for tyres is also considered. Emissions associated with cultivation operations include diesel combustion (accounted for within modelled diesel use) and emissions to soil from tyre abrasion, also modelled using ecoinvent processes. For the latter, this is considered only for four-wheel tractors.

2.1.7.2.8 Crop transport

The transport of crops is estimated using information from farmers about their points of sale. For conventional farmers supplying to retail chains, transport to regional distribution centres (RDCs) and then transport to individual retail locations are both considered. For the prior, specific information has been provided by farmers regarding the locations of RDCs supplied to and the relevant proportions of crops supplied. However, distances of RDCs to shops are based mostly on estimates, using information that some farmers were able to provide from supermarket chains, as well as estimating mid-point distances between different RDC sites in the region or country of interest. An additional consideration is also required for UK organic farms that distribute crops through home-delivered ‘veg boxes’; in this case, specific information about the delivery routes has been modelled based on the farmer or veg box distributor.

Transport is modelled using ecoinvent v.3.7.1 processes, which include burdens of fuel use as well as vehicles. All vehicles have been assumed as EURO4 for consistency, although in reality this likely varies. Vehicle or transport type varies among farms and is defined by farmers. Lorries or refrigerated lorries are used for all conventional farms; for transport via these types, ecoinvent processes define outputs based in kg*km. Thus, this has been modelled using an average one-way transport distance, based on the weighted average of all sale points, multiplied by the total sellable crop (e.g., after subtracting processing waste from harvested crop). One-way transport distances are considered for this process because ecoinvent freight transport processes are calculated for an average load factor, and thus include empty return trips already.

However, all organic farms transport crops either in passenger cars, petrol vans, or diesel transit vans; for the latter case, this can be modelled in terms of kg*km within ecoinvent like for lorry transport. However, when passenger cars or petrol vans are used, transport must be modelled using passenger car processes, which are listed in terms of km. In this case, the total round-trip distance travelled by all the kale or tomato crops, to all the points of sale, must first be calculated (i.e., trip distance km * total number of trips for which the crop is included). This is then allocated to the crop of interest based on the proportion of this crop out of all other crops being transported. This is performed using mass allocation or another estimate provided by the farmer if mass estimates could not be provided (e.g., total number of crops).

2.1.7.2.9 Infrastructure

The major types of infrastructure considered in this LCA include: polytunnels, greenhouses, and glasshouses used for seedling production and crop cultivation; sheds used for housing agricultural equipment; packhouses on or near farms, used for packaging and processing crops from the farm; insulated or cold storage facilities (e.g., walk-in coolers or cooled shipping containers), used for storing the crop; and regional distribution centres (RDCs), used for storing the crop temporarily before redistribution to specific retail sites.

Polytunnels and greenhouses have been modelled using material amounts (in kg m^{-2}) provided by relevant suppliers for each type of polytunnel used. These LCIs are provided in Section 2.1.7.5.1. However, large-scale glasshouses, such as those used by nurseries in the UK and for UK conventional tomato production, are modelled using AGRIBALYSE v.3.0 processes.

The use of shed space for agricultural equipment is modelled within crop cultivation LCIs, based on information provided by farmers about the type and size of equipment storage spaces. Shed infrastructure was modelled mainly using AGRIBALYSE v.3.0 or WFLDB v.3.5 processes, depending on which was more relevant. Most farms had very simple wooden shed or barn structures with bare or cement ground, for which AGRIBALYSE v.3.0 shed processes were more suitable. Shed space was generally allocated to the crop of interest based on the weight harvested over the year out of the total weight of crops harvested (mass allocation). Where farmers could not estimate a space used, but highlighted that this infrastructure did exist on the farm, then it has been modelled based on allotted shed space for agricultural equipment listed in ecoinvent processes; in this case, the shed space has not been listed explicitly within the cultivation LCIs. In other cases, farms may not utilise any shed space to house agricultural equipment, instead storing equipment outside, in which case no infrastructure is considered.

Packhouses for storage and packaging, which are almost always on or near the farm site, have also been modelled. Packhouse infrastructure varied on farms. Generally, organic farms had no packhouses or very simple wooden structures. U.S. conventional farms all had simple wooden barn structures as packhouses, whilst all UK conventional farms had metal-framed buildings and warehouses. Thus, for the prior case, AGRIBALYSE v.3.0 shed processes were used to model wooden barns; for the latter case, ecoinvent v.3.7.1 processes for buildings were used. As for shed infrastructure, packhouse infrastructure was normally allocated to the crop of interest using mass allocation, unless another estimate was provided by the farmer.

Several types of cold storage facilities are used by farmers. These include insulated cool rooms (walk-in coolers) that are cooled using air conditioning (A/C) units; cool rooms using conventional refrigeration; and standing refrigerators. For all cold storage facilities, space is allocated to the crop of interest based on the space the crop takes up in the cold storage area and the time that this space is used for the crop over the year; again, this is based on primary data from farmers. In the few cases where farmers were unable to provide an estimate of this, space use within cool rooms (walk-in coolers) has been estimated based on the size of the boxes or crates used to store the product, multiplying this by a factor of three for chilled food products as per the Product Environmental Footprint Category Rules (PEFCR) Guidance (European Commission, 2018). Further information on the processes used for cold storage infrastructure is provided in Section 2.1.7.5.2.

In the U.S., many organic farms use insulated cold storage rooms (walk-in coolers) that are cooled by air conditioning (A/C) units rather than conventional refrigeration units. This is facilitated by the use of CoolBot© systems (https://www.storeitcold.com/). Normally, A/C units cannot achieve temperatures below 15.50°C/60°F; however, with the use of CoolBot systems linked to A/C units, insulated rooms can be cooled to temperatures of 2.20°C/36°F. These systems thus provide farms with a low-cost way to implement cold storage facilities on site, and are thus commonly used across small farms in the U.S. For farms using these systems, infrastructure has been modelled using secondary databases based on the structures used to create the walk-in cooler / cold storage facility. A/C units and relevant refrigerant emissions have also been modelled. Details on the modelling of infrastructure and A/C units are provided in Section 2.1.7.5.2. The energy use from these cooling systems has generally been estimated based on information provided in the 2009 report of CoolBot-powered walk-in coolers (CDH Energy Corp, 2009). Estimations were made for specific farms based on the proportional size/capacity of their cool rooms and A/C units vs. those analysed in the study; however, these were generally similar.

Other cold storage facilities modelled use conventional refrigeration, either as walk-in coolers or shipping containers. Energy use from cold storage is either estimated based on figures provided by farmers (often based on total energy bills or usage), or, where farmers were unable to provide an estimate, using approximations for cold storage rooms provided in PEFCR guidance (European Commission, 2018). Finally, some small farms in the U.S. utilised standing refrigerators for crop storage. Energy use for these has been estimated mainly using wattage ratings and times of use provided by farmers. Details on the modelling of infrastructure and relevant emissions for all cold storage types is provided in Section 2.1.7.5.2.

Finally, distribution centre infrastructure and associated energy, water, and refrigerant use has also been modelled based on PEFCR Guidance (European Commission, 2018). This assumes 30 kWh $(m^2a)^{-1}$ operational energy use, 360 MJ operations natural gas heating $(m^2a)^{-1}$, 12.2 L water use $(m^2a)^{-1}$, and an additional 40 kWh $(m^3a)^{-1}$ electricity use per volume of cold stored product. RDCs are considered only for conventional farms supplying to supermarkets and wholesalers. The RDC structure itself is modelled using ecoinvent building processes. Further detail on the modelling of infrastructure for these spaces is provided in Section 2.1.7.5.3.

2.1.7.2.10 Additional exclusions

This section provides further detail on items excluded from LCIs that have not yet been dictated in the prior sections. In accordance with PAS 2050:11 specifications for the lifecycle greenhouse gas emissions of goods and services, the following items are not considered: human energy inputs / human labour (e.g., from harvesting); transport of employees to and from their normal places of work; and transport of consumers to points of sale. Additional excluded items are:

- Crop seeds, due to limited information about seed production, especially for organic seeds; however, cover crop seeds have been considered and modelled using ecoinvent processes
- All energy uses associated with the retail environment, as this is considered out of scope
- Any additional packaging for crops that may occur after leaving farm packhouses, due to limited information; however, in most cases it is assumed that all packaging will occur at the farm packhouses
- Packhouse consumables (e.g., gloves, hairnets, trash bags)

2.1.7.3 Kale LCIs

This section provides an overview of the resources and material flows modelled for kale lifecycles on case study farms in the U.S. and UK. This includes data from nine case study farms in the U.S., including one urban organic farm (U-O), four peri-urban organic farms (PU-O), three rural organic farms (R-O), and one rural conventional farm. Twelve case study farms are considered in the UK, which include one urban organic farm, four peri-urban organic farms, two rural organic farms, and five rural conventional farms.

Lifecycle inventories (LCIs) are provided for each individual farm, separated based on country. These are also listed separately for the different phases of the food supply chain included within the system boundaries of this LCA, specifically: seedling production; crop cultivation; on-farm processing, packaging, and storage; and distribution to final points of sale. This is designated in the following sections.

2.1.7.3.1 Seedlings

Lifecycle inventories for the production of kale seedlings are provided in Table 4 through Table 6 for U.S. kale farms and Table 7 through Table 9 for UK kale farms. In these sets of tables, the first provides an overview of resource flows, specifying the number of seedlings germinated, where this takes place (e.g., on the farm or at a nursery), the infrastructure used, and energy and water use. The second provides an overview of the growing media or potting mix components used and any fertilisers or pesticides applied, including organic fertilisers. Finally, the third table lists all non-infrastructural material uses. These materials have been modelled using ecoinvent v.3.7.1, with the specific manufacturing processes used also provided in these tables. Material lifetimes as designated by the farmer or supplier are also included, listed in parentheses next to amounts used. All material amounts listed have already been allocated based on the time of use for the crop lifecycle and the lifetime.

Farms or nurseries will generally over-sow seeds during germination to ensure that an adequate amount of seedlings is produced; thus, the total number of seedlings germinated is not necessarily equal to the number of seedlings that will ultimately be transplanted in the field. All materials and resources are listed in terms of the number of seedlings that are ultimately *transplanted*, which is the output of these inventory models.

Overall, it can be seen that the U.S. organic farms tend to use a wider range of growing media and organic fertilisers during seedling production, while UK organic farms tend to rely on the use of compost as the main fertility source. Additionally, all U.S. farms germinate seedlings in polytunnels or low tunnels, while some farms in the UK, especially the larger-scale nurseries, tend to use glasshouses for production. The U.S. conventional farm also produces all seedlings on site, in comparison to the UK conventional kale farms, which all purchase in seedlings from nurseries. Heat is generally not used for kale germination.

Table 4 – Resource flows for U.S. kale seedlings

Flows	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-O-3	R-C-3
Seedlings germinated	Number	832	1,536	256	1,224	408	4,456	2,048	1,200	6,608,800
Seedlings planted	Number	749	990	230	1,200	360	4,233	2,000	1,200	6,008,000
Total germination time	Weeks	14	6	6	6	12	15	15	8	6 [b]
Germination time per round	Weeks	7	6	6	6	4	5	5	4	6
Polytunnel type [a]	Type	US-P-1	US-P-2	US-P-2	Low tunnel	US-P-4	US-P-1	US-P-1	US-P-1	US-P-2
Germination location	Type	On farm	Nursery	Nursery	On farm	On farm	On farm	On farm	On farm	On farm
Transport from nursery	km, one way	0	371	372	0	0	0	0	0	0
Resources (seedling transplanted)$^{-1}$										
Area use [b]	m^2	0.0024	0.0018	0.0013	0.0019	0.0022	0.0061	0.0019	0.0062	0.0070
Energy use, GA mix	kWh	0.0004	0.004	0.003	0	0.26	0.007	0.015	0.068	0.003
Energy use, solar	kWh	0	0	0	0	0	0	0	0.063	0
Propane use	kg	0	0.004	0.003	0	0	0	0	0	0.009
Water use	L	0.37	0.10	0.071	0.093	0.063	0.302	0.350	0.114	0.040

[a] See Table 40 for polytunnel infrastructure LCIs. Note that these are only listed for labelled infrastructures. Low tunnel materials are included within the seedling material use LCI, in Table 6.
[b] Total germination time is actually longer for these conventional farms, as they use a staggered seeding model; however, detailed information was not provided about how long the total germination time was, so burdens have been estimated based on the average germination time per seedling.
[b] This is the area used over the total germination time, per seedling transplanted.

Table 5 – *Fertiliser, growing media, and pesticide inputs for U.S. kale seedlings, in g (seedling transplanted)$^{-1}$*

Inputs	**U-O-1**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-O-3**	**R-C-3**
Sand	0	0	0	1.34	0	0	1.17	0	0
Peat moss	4.37	6.33	4.54	0.71	2.60	1.06	1.18	1.32	5.41
Perlite / Vermiculite	2.19	1.06	0.78	3.35	0.31	1.02	0	0.41	0.52
Wood chips	0	0	0	22.8	0	0	0	0	0
Lime	0.08	0	0	0	0.14	0.025	0	0	0.067
Compost, purchased	17.5	0	0	0	0	19.5	8.28	0	0
Worm castings	0	1.08	0.78	0	0	0	0	0.18	0
Azomite	0.025	0	0	0	0	0.051	0	0	0
Blood meal / feathermeal	0.051	0	0	0	0	0.18	0	0	0
Kelp meal / seaweed emulsion	0.025	0	0	0	0.003	0	0	0	0
Fish emulsion	0	0.84	0.600	0	0.003	0	0	0	0
Alfalfa meal	0.10	0	0	0	0	0	0	0	0
Other organic fertiliser mixes	0	0	0	3.02	0	0	0	0	0
Mineral NPK fertilisers	0	0	0	0	0	0	0	0	0.28
N, in all fertilisers	0.19	0.059	0.042	0.009	0.0002	0.22	0.084	0.0004	0.037
P_2O_5, in all fertilisers	0.042	0.025	0.018	0.003	0.0001	0.049	0.021	0	0.006
K_2O, in all fertilisers	0.053	0.034	0.024	0.003	0.00004	0.057	0.024	0	0.037
Pesticides (g ai) [a]	0	0	0	0	0	0	0	0	0
Compost transport (km) [b]	282	0	0	0	0	214	813	0	0
Other input transport (km) [b]	14.0	167	167	8.69	31.2	295	813	9.65	1,347

[a] These are the pesticides applied, in grams active ingredient (ai) per seedling transplanted.
[b] This is the average one-way distance that compost and other inputs travel from the place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops.

Table 6 *– Material use for U.S. kale seedlings, in g (transplanted seedling)$^{-1}$*

Items	**Material**	**Process**	**U-O-1**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-O-3**	**R-C-3**
Pots and trays	PS	Injection moulded	0.009 (2) [a]	0.77 (1)	0.55 (1)	0.29 (3)	0	0.066 (5.6)	0	0.12 (4)	0
Trays	PS	Foaming	0	0	0	0	0	0	0	0	0.08 (7)
Long-life trays	Acrylic	Injection moulded	0.081 (30)	0	0	0	0	0	0.064 (30)	0	0
Humidity domes	PET	Injection / blow moulded	0	0	0	0	0	0	0	0.052 (3.5)	0
Low tunnel, film	PE	Extrusion, film	0	0	0	0.020 (6)	0	0	0	0	0
Low tunnel, hoops	PVC	Extrusion, pipe	0	0	0	0.003 (40)	0	0	0	0	0
Material transport to farm (km) [b]	n/a	n/a	633	75	75	235	31	678	262	901	75

[a] Material amounts are provided in g (transplanted seedling)$^{-1}$, allocated based on time of use and material lifetime. Material lifetimes, in years, are provided in parentheses (), but note that these have already been applied to the provided material amounts.
[b] This provides the average one-way distance that materials are transported from their place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops. However, these distances are accounted for within modelled ecoinvent processes.

***Table 7** – Resource flows for UK kale seedlings*

Flows	Unit	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Seedlings germinated	Number	2,640	520	1,300	832	4,140	2,400	7,992	12,000	104,328	1,370,000	2,750,000	4,400,000
Seedlings planted	Number	2,500	420	900	749	4,000	2,400	7,800	12,000	104,328	1,233,000	2,667,500	3,960,000
Total germination time	Weeks	6	3	6	10	4	5	5	7.5 [b]	11 [b]	8 [b]	8.7 [b]	8.9 [b]
Germination time per round	Weeks	6	3	6	5	4	5	5	7.5	11	8	8.7	8.9
Polytunnel / glasshouse type [a]	Type	UK-P-1	UK-G-1	UK-P-3	Reclaimed tunnel	Glasshouse	Glasshouse	Glasshouse	Glasshouse	Glasshouse	Glasshouse	US-P-2	Glasshouse
Germination location	Type	On farm	On farm	On farm	On farm	Nursery	Nursery	Nursery	Nursery	Nursery	Nursery	Nursery	Nursery
Transport from nursery	km, one way	0	0	0	0	1.5	330	290	202	106	7.2	40.1	3
Resources (seedling transplanted)$^{-1}$													
Area use [c]	m^2	0.0013	0.0035	0.0050	0.0015	0.0007	0.0046	0.0047	0.0008	0.0015	0.0007	0.0009	0.0014
Electricity use	kWh	0	0	0	0	0	0.0005	0.0005	0.00001	0.0013	0.0005	0	0.0011
Propane use	kg	0	0	0	0	0	0	0	0	0	0	0	0.00002
Water use	L	0.30	0.24	0.23	0	0.11	0.009	0.009	0.11	0.17	0.19	0.17	0.11
Water use, rainwater	L	0	0	0	0.20	0	0	0	0	0	0	0	0

[a] See Table 40 for polytunnel and glasshouse infrastructure LCIs. Note that these are only listed for labelled infrastructures. Those listed as just 'glasshouse' are modelled using the AGRIBALYSE v.3.0 database.
[b] Total germination time is actually longer for these conventional farms, as they use a staggered seeding model; however, detailed information was not provided about how long the total germination time was, so burdens have been estimated based on the average germination time per seedling.
[c] This is the area used over the total germination time, per seedling transplanted.
[d] Water use from captured rainwater is listed here for information purposes, but note that this does not contribute to water consumption impacts in the impact assessment.

Table 8 *– Fertiliser, growing media, and pesticide inputs for UK kale seedlings, in g (seedling transplanted)$^{-1}$*

Inputs	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-1**	**R-C-2**	**R-C-3**	**R-C-4**	**R-C-5**
Sand	0	0	0	3.48	0	0	0	0	0	0	0	0
Peat	0	0	0	1.42	0	0	0	0.33	0.97	0.43	0.51	1.18
Vermiculite	0	0	0	0	0	0	0	0.063	0	0	0	0
Coconut coir	1.85	5.22	0	1.90	1.83	2.61	2.70	0.74	0	0	0	0
Compost, purchased	13.4	5.50	0	0	1.93	2.75	2.85	0	0	0	0	0
Compost, homemade	0	0	13.9	0	0	0	0	0	0	0	0	0
Wood chips / forest by-products	0	0	0	5.68	0	0	0	0	0	0	0	0
Other organic fertiliser mixes	0	0.37	0	0	0	0	0	0	0	0	0	0
Mineral NPK fertilisers	0	0	0	0	0	0	0	0.134	0	0	0.34	0.07
N, in all fertilisers	0.005	0.015	0.113	0.004	0.006	0.009	0.009	0.021	0	0	0.001	0.008
P_2O_5, in all fertilisers	0.003	0.007	0.050	0.001	0.004	0.006	0.006	0.031	0	0	0.003	0.030
K_2O, in all fertilisers	0.020	0.022	0.092	0.014	0.008	0.011	0.012	0.021	0	0	0.003	0.008
Pesticides (g ai) [a]	0	0	0	0	0	0	0	0.011	0	0.010	0	0.011
Compost transport (km) [b]	87	0	0	0	914	665	665	0	0	0	0	0
Other input transport (km) [b]	87	539	0	153	914	665	665	244	215	55	75	68

[a] These are the pesticides applied, in grams active ingredient (ai) per seedling transplanted.
[b] This is the average one-way distance that compost and other inputs travel from the place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops.

Table 9 *– Material use for UK kale seedlings, in g (transplanted seedling)$^{-1}$*

Items	Material	Process	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Pots and trays	PS	Injection moulded	0.18 (16) [a]	0.12 (8)	0	0	0	0.79 (1)	0	0	0	0	0	0
Pots and trays	PP	Injection moulded	0	0	0	0.095 (3.5)	0.034 (12.5)	0.20 (5)	0.31 (7.5)	0.12 (7)	0.24 (12.5)	0.067 (7.5)	0.045 (12.5)	0.101 (12.5)
Fleece	PP	Extrusion, film	0	0.00007 (7.5)	0	0	0	0	0	0	0	0	0	0
Crates	HDPE	Injection moulded	0	0	0	0	0	0	0	0.23 (12.5)	0	0	0	0
Plastic liners	PE	Extrusion, film	0	0.0012	0	0.0074 [b] (17.5)	0	0	0	0	0	0	0	0
Hoops bench cover	PVC	Extrusion, pipe	0.0027 (40)	0	0	0	0	0	0	0	0	0	0	0
Stillages	Steel	Section bar rolling	0	0	0	0	0	0	0	0	0	0	0.0049 (10)	0
Benches & stillages	Wood	Sawnwood	0.07 (20)	0.015 (20)	0	0	0	0	0	0.022 (5)	0	0.016 (5)	0	0.033 (5)
Benches (filler)	Sand	n/a	0	0.078	0	0	0	0	0	0	0	0	0	0
Material transport to farm (km) [c]	n/a	n/a	189	194	0	94	75	91	91	75	75	75	75	75

[a] Material amounts are provided in g (transplanted seedling)$^{-1}$, allocated based on time of use and material lifetime. Material lifetimes are provided in parentheses (), and listed in years unless otherwise specified; note that these have already been applied to the provided material amounts.
[b] This is the plastic cover for the polytunnel on this farm. This input is listed here, instead of in the polytunnel infrastructure LCIs, because this farm uses a polytunnel made of all reclaimed materials, except for the plastic sheeting, which is the only purchased input.
[c] This provides the average one-way distance that materials are transported from their place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops. However, these distances are accounted for within modelled ecoinvent processes.

2.1.7.3.2 Cultivation

This section depicts the main resource and material flows during kale cultivation. For details about the assumptions made when modelling these material flows, refer to Section 2.1.7.2.

Table 10 and Table 13 provide an overview of the main characteristics and resource flows for kale cultivation on farms in Georgia, USA and England, UK, respectively. In both cases, kale is cultivated mainly in open fields. Some organic farms may also cultivate a portion of kale, or all kale, in polytunnels, but this is less common than for tomatoes. Although kale is a biennial crop, it is mainly grown as an annual among all farms studied, and only annual kale crops are considered for this LCA. The gross area used for kale cultivation on each farm is designated as the full polytunnel or field size, including crop rows, aisles, field margins, and roadways. Gross area is more commonly considered by growers in the UK when describing inputs or yields. Cropped area, as a percent of the total gross area, is also listed; this mainly refers to the actual cropped rows or beds. Growers in the U.S. tend to describe inputs and yields based on cropped area. However, inputs within the cultivation LCIs are mainly listed in terms of gross area unless otherwise stated, to maintain consistency. Inputs per cropped area can be calculated by multiplying by the listed cropped area percent for any given farm.

The main growing season for U.S. conventional kale production takes place from August through May, with harvest lasting for 7-8 months and finishing when conditions become too hot for the crop. In the UK, kale seedlings are typically transplanted from March-May and harvested until January or February, with harvest typically lasting for 7-10 months. For all farms, kale seedlings are germinated separately and then transplanted into the field, but some organic farms in the U.S., as well as both conventional and organic farms in the UK, also direct seed a portion of their crop; this usually occurs later in the season after the 'main crop' to extend harvest. Indeed, a wide variety of farms both in the U.S. and UK generally stagger kale plantings or perform 2-3 main plantings throughout the year to extend the harvest season. This is apparent in Table 10 and Table 13, when the listed harvest season (in weeks) exceeds that of the average growing cycle per kale planting.

Harvesting practices for kale also vary across farms. Most organic farms take several harvests from kale, as new leaves will re-grow for some time after an initial harvest; this is also done by the conventional U.S. farm, which takes 3-4 harvests from the same crop, as well as some of the UK conventional farms. However, some of the UK conventional farms also harvest the crop to completion, i.e., only take one harvest. In all cases, human labour is used for harvesting, although this may be used in conjunction with tractors and machinery on conventional farms.

Harvest waste amounts range widely across farms, anywhere from 0-75%. The highest levels of harvest waste are generally seen on organic farms. Reasons for waste include pest damage, appearance issues, and inability to fully harvest the crop (e.g., lack of labour); additionally, in some cases crop was left unharvested if it was assumed that it would not be sold. This particularly contributed to the high harvest waste seen for US-PU-O-3.

Regarding resource use, most UK kale production can take place with minimal or no irrigation, seen as the majority of UK kale farms have 0 water use for irrigation in Table 13; however, in the U.S., almost all farms irrigate kale crops, including the conventional farm, which utilises drip irrigation for this purpose. Thus, any listed electricity use during cultivation is mainly a result of energy used for irrigation. No farms use heating for their kale crop; however, there is some propane use on U.S. organic farms, due to the application of flame weeding. Listed diesel use within Table 10 and Table 13 is mainly from tractor operations on farms, as well as from any vehicle use to transport workers or crops between fields. Petrol may also be used for certain two-wheel tractors and for on-farm vehicle transport, mainly in the U.S. The use of shed space for agricultural equipment, allocated to kale, is also considered, although in some cases no shed space is used if farmers store all equipment outside, or if this has been alternatively modelled within ecoinvent machinery processes because farmers could not provide an estimate.

Fertility and pesticide inputs are provided in Table 11 and Table 14 for U.S. and UK farms, respectively. These are listed in terms of the weight applied out of the gross area used for cultivation. In terms of fertility inputs, generally lower amounts of nutrient inputs are required for kale than tomato crops. Organic farms in the U.S. tend to use a wider range of inputs than in the UK, where mainly compost or manure is used. Cover crops are utilised on farms in both the U.S. and UK to provide nutrients and also for mulching around the kale crop. Total N inputs from cover crop residues as allocated to kale is listed, calculated using equations and information provided in Section 2.1.9.2.2. Total N, soluble N, P_2O_5, and K_2O inputs from all other fertilisers are also listed.

For pesticide inputs, it can be seen that organic farms in the U.S. tend to apply various biological and other organic pesticides (e.g., from natural sources), whereas this was not seen on organic farms in the UK. Conventional kale production in the U.S. generally includes soil fumigation as a main practice, performed just before plastic is laid over the crop rows. This is not a common practice in the UK; however, molluscicides are applied more commonly for conventional kale production in the UK, which is not seen for the conventional farm in Georgia.

Finally, the material inputs and cultivation machinery used for kale production is provided in Table 12 and Table 15 for U.S. and UK farms, respectively. Material amounts are provided in terms of gross kale production area and are already allocated based on time of use out of the year and material lifetime. However, material lifetimes (in years) have also been included within the table for informational purposes. Reclaimed materials have been provided in these tables and designated with a footnote.

The main material uses for kale production are from plastic mulches, netting or fleeces, and irrigation materials. Plastic mulch laid under the crop is more common for U.S. cultivation, whilst in the UK, fleece or other mesh nettings are often used to cover the crop to prevent pests and frost. Smaller farms in the U.S. and UK may also make low tunnels over the crop, using steel wire hoops and a fleece cover. In some cases, these fleeces may be weighed down with other materials; for example, UK-PU-O-3 uses reclaimed tyres to weigh down netting materials. Other materials inputs include drip irrigation materials (such as drip tape and piping), which again is more commonly seen for U.S. farms since most UK farms do not irrigate kale. Rainwater harvesting equipment, such as tanks and pond liners, are also included, mainly seen for UK organic farms. Finally, any shipping containers used for equipment storage are listed, as well as tractors and implements (e.g., ploughs, harrows), as allocated to the kale crop.

Table 10 – U.S. kale cultivation characteristics and major resource flows

Flows	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-O-3	R-C-3
Gross area	m^2	100	70	111	93	111	747	525	446	1,618,743
Percentage as cropped area	%	67%	62%	73%	100%	69%	73%	80%	55%	85%
Average growing cycle [a]	weeks	16.0	14.3	19.8	18.4	28.9	20.8	34.6	36.3	21.7
Total harvest season [a]	weeks	26.0	13.0	26.0	10.8	43.3	36.8	32.5	52.0	32.5
Production in field	%	100%	100%	42%	100%	100%	80%	42%	25%	100%
Production inside	%	0%	0%	58%	0%	0%	29%	58%	75%	0%
Polytunnel type [b]	Type	n/a	n/a	US-P-3	n/a	n/a	US-P-4	US-P-1	US-P-1	n/a
Planting density	Plants (cropped m^{-2})	2.15	14.2	3.41	12.9	3.23	5.66	3.81	38.6	3.71
Produced crop [c]	kg (gross m^{-2})	1.53	1.95	0.95	2.94	1.99	1.82	1.04	0.79	2.19
Harvested crop [c]	kg (gross m^{-2})	1.45	1.95	0.76	0.75	1.81	0.72	1.04	0.71	2.19
Harvest waste [c]	%	5.6%	0%	20% [d]	75%	9.2% [d]	60.6%	0%	10% [e]	0%
Resources (gross m^{-2})										
Electricity use, GA mix	kWh	1.09	0	0	0	0	0.474	0.032	0.988	0.025
Electricity use, solar	kWh	0	0	0	0	0	0	0	0.920	0
Diesel use	L	0.006	0	0	0	0	0.018	0.006	0.019	0.009
Petrol use	L	0	0.024	0.659	0.012	0	0.002	0	0	0
Propane use	kg	0	0	0.0348	0	0.0271	0	0	0	0
Irrigation water use	L	154	236	59.3	0	203	126	89.6	306	203
Pesticide / spray water use	L	1.02	0	0	0	0.209	0.041	0.087	0.815	0.466
Shed area, allocated [f]	m^2a	0	0	0	0.002	0.003	0.015	0.008	0	ecoinvent

[a] The average growing cycle per planting is calculated with a weighted average of all plantings, based on area. The total harvest time may be longer than the average planting time if plantings are staggered.

[b] See Table 40 for LCIs for polytunnel infrastructures.

[c] Produced crop refers to the total crop that was grown in the field, based on farmer estimates or measurements. Harvest is the amount of crop that was harvested from the field. Harvest waste percent is then calculated as the total weight of crop wasted during harvest (i.e., sorted out in the field) divided by the total produced crop. All harvest waste is assumed to be returned to the soil unless otherwise denoted.

[d] This harvest waste is composted; [e] this harvest waste is fed to animals on the farm.

[f] The shed area used to house cultivation equipment has been allocated to kale based on time of use and / or weight or area allocation, depending on which is more appropriate. Note that the amount of shed space modelled here is only that specified by the farmer; when the farmer was unable to provide an estimate of this, shed space has been modelled using ecoinvent databases (listed as 'ecoinvent') and is not reproduced here.

Table 11 *– Fertility and pesticide inputs for U.S. kale cultivation*

Inputs per gross m^2 cultivated area	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-O-3	R-C-3
Cover crop area, allocated to kale	$m^2a\ m^{-2}$	0.15	0	0	0	0	0.25	0	0	0.14
Cover crop type (% by seed weight in mix)	type	40% oat 40% pea 20% rye	n/a	n/a	n/a	n/a	50% rye / clover 50% cowpea / wheat / sunflower	n/a	n/a	60% wheat 40% rye
Cover crop seed per area of cover crop, allocated to kale	$kg\ m^{-2}$	0.007	0	0	0	0	0.036	0	0	0.006
Compost, purchased	$kg\ m^{-2}$	19.5	8.68	5.72	0	0	1.05	0	0	0
Compost, homemade	$kg\ m^{-2}$	0	0	0	0.014	0	0	0	0	0
Dried, pelleted manure	$kg\ m^{-2}$	0	0.081	0.015	0	0	0	0	0	0
Feather meal	$kg\ m^{-2}$	0.002	0	0	0	0	0.051	0.117	0.102	0
Kelp / seaweed meal & fish emulsion	$kg\ m^{-2}$	0.009	0	0	0	0.003	0	0	0.040	0
Other plant-based meals	$kg\ m^{-2}$	0	0	0	0	0	0.052	0.117	0	0
Peat moss	$kg\ m^{-2}$	0	0	0	0	0	0.289	0	0	0
Lime, as $CaCO_3$	$kg\ m^{-2}$	0.014	0	0	0	0	0	0	0	0
Other organic fertiliser mixes	$kg\ m^{-2}$	0.009	0.016	0	0	0	0.003	0	0	0
Mineral NPK fertilisers	$kg\ m^{-2}$	0	0	0	0	0	0	0	0	0.450
Total N, from cover crop residues	$g\ m^{-2}$	2.91	0	0	0	0	9.13	0	0	1.34
Total N, from fertilisers	$g\ m^{-2}$	197	91.7	58.4	0.231	0.127	19.7	18.7	15.2	67.9
Soluble N, from fertilisers	$g\ m^{-2}$	0.413	4.76	2.75	0.012	0.100	0.581	1.07	2.35	67.9
P_2O_5, from fertilisers	$g\ m^{-2}$	48.7	25.0	14.9	0.091	0.08	8.71	0.583	0.399	3.22
K_2O, from fertilisers	$g\ m^{-2}$	56.5	27.6	17.0	0.164	0.023	11.2	2.92	0.399	35.3
Biological pesticides, active ingredient (ai)	$g\ ai\ m^{-2}$	0	0	0	0	0	0.005	0	0.072	0
Other organic pesticides, active ingredient (ai)	$g\ ai\ m^{-2}$	1.83	0	6.31	0	0.114	0.001	0.225	41.5	0
Pesticides, active ingredients (ai)	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0	10.08
…Insecticides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0	0.158
…Herbicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0	0.152
…Fungicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0	0.955
…Fumigants	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0	8.82
Compost transport [a]	km	282	3	241	0	0	340	0	0	0
Other input transport [a]	km	574	650	773	13	31	540	738	700	135

[a] This is the average one-way distance that compost and other inputs travel from the place of sale / collection to the farm. This value does not include other transport, such as from manufacturing centres to shops.

__Table 12__ – Materials and equipment for U.S. kale cultivation, in g (gross m^{-2})

Item	Material	Process	U-O-1	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-O-3	R-C-3
Shadecloth / landscape fabric	HDPE or PP	Extrusion and weaving	18.3 (4)	0	2.32 (9.2)	0	0	4.53 (12)	0	1.63 (20)	0
Plastic mulch / tarps	LDPE	Extrusion, film	23.4 (3)	3.48 (10)	0	0	0	0	0	0	4.64 (4 uses)
Row cover / fleece	PP	Spunbond	0	0	0	0	2.14 (3)	0	0.150 (5.5)	0.706 (2)	0
Low tunnel hoops	Fibreglass	Hand lay-up	0.501 (2)	0	0	0	0	0	0	0	0
Low tunnel hoops	Steel	Wire drawing	0	0	0	0	5.02 (13)	0	0	0.177 (3)	0
Drip tape	LDPE	Extrusion, film	2.85 (2.5)	0	0	0	0	2.70 (10)	4.06 (4)	8.62 (2)	2.60 (4 uses)
Drip irrigation mainline	LLDPE	Extrusion, pipe	0.447 (4)	2.16 (6)	0	0	0	0.052 (20)	0.097 (15)	0	0.180 (10)
Drip irrigation mainline	PVC	Extrusion, pipe	0	0.140 (40)	0	0	0	0	0	0.253 (40)	0
Sprinklers	Acetal	Injection moulded	0.002 (6)	0	0	0	0	0	0	0	0
Sprinklers	PVC + Brass + PE	Injection moulded / cast metal	0	0	0	0.119 (6)	0	0	0	0	0
Harvest crates	PP or HDPE	Injection moulded	2.98 (10)	0	0.26 (7)	0.305 (10)	0.072 (15)	0.027 (20)	1.50 (2.5)	0.290 (5)	0
Shipping containers for equipment store	Steel	Metal working, welding	11.5 (40)	0	0	0	0	0	0	0	0
Tractor, 2-wheel [b]	Mostly steel	Varies	0.225 (3,000 hrs)	0.19 (3,000 hrs)	0	0.130 (3,000 hrs)	0	0	0	0	0
Tractors, small (<50 HP) [b]	Mostly steel	Varies	0	0	0	0	0	0.702 (9,000 hrs)	0.221 (9,000 hrs)	1.98 (9,000 hrs)	0
Tractors, large (>100 HP) [b]	Mostly steel	Varies	0	0	0	0	0	0	0	0	0.47 (9,000 hrs)
Implements [b]	Mostly steel	Varies	0	0	0	0	0	0.938 (varies)	1.026 (800 hrs)	3.61 (800 hrs)	0.49 (varies)

[a] All material amounts are provided in g (gross m^{-2}) and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated.
[b] Tractors are designated as 'small' or 'large' based on horsepower (HP) rating. All tractors, implements, and other cultivation equipment are modelled using ecoinvent processes and thus specific material components are not provided in accordance with copyright principles. Amounts have been allocated to kale based on the time of use for the crop, out of the total equipment lifetime, in hours. Lifetimes are estimated first from farmer data, then from secondary databases and literature sources.

Table 13 – UK kale cultivation characteristics and major resource flows

Flows	Unit	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Gross area	m^2	765	184	302	33.4	900	420	2490	6937	21,100	880,000	943,000	1,974,500
Percentage as cropped area	%	80%	87%	76%	67%	80%	67%	79%	62%	90%	92%	85%	88%
Average growing cycle [a]	weeks	26	43	39	30	43	22	37	36	48	42	42	17
Total harvest season [a]	weeks	26	35	30	27	30	13	30	35	39	52	43	30
Production in field	%	100%	100%	100%	100%	100%	0	100%	100%	100%	100%	100%	100%
Production inside	%	0	0	0	0	0	100%	0	0	0	0	0	0
Polytunnel type	type	n/a	n/a	n/a	n/a	n/a	UK-P-1 [b]	n/a	n/a	n/a	n/a	n/a	n/a
Planting density	plants (cropped m^{-2})	4.35	4.08	2.63	3.91	33.6	5.56	8.54	3.99	2.78	5.49	2.20	2.83
Produced crop [c]	kg (gross m^{-2})	0.523	1.85	0.629	2.74	3.15	2.21	0.637	0.937	1.24	1.35	0.479	0.963
Harvested crop [c]	kg (gross m^{-2})	0.261	0.924	0.510	2.15	0.787	2.07	0.512	0.865	0.933	1.14	0.479	0.861
Harvest waste [c]	%	50%	50%[d]	19%[d]	22%[d]	75%	6%	20%[d]	8%	25%	15%	0	11%
Resources (gross m^{-2})													
Electricity use, GB mix	kWh	0	0	0	0	0	0.006	0	0.140	0.001	0	0	0
Electricity use, solar	kWh	0	0	0	0	0	0	0	0	0.0003	0	0	0
Diesel use	L	0.057	0.031	0.008	0	0.053	0.005	0.036	0.071	0.206	0.030	0.018	0.030
Petrol use	L	0	0.0006	0.005	0	0	0	0.0002	0	0	0	0	0
Irrigation water	L	0	4.57	4.47	0	0	130	1.57	31.6	36.0	0	0	0
Irrigation, rainwater [e]	L	0	0	0	2.99	0	130	0	0	0	0	0	0
Pesticide / spray water use	L	0	0	0	0	0	0	0	0.112	0.178	0.152	0.085	0.053
Shed area, allocated [f]	m^2a	0	ecoinvent	0	0	0	0	0.002	0.002	ecoinvent	ecoinvent	ecoinvent	0

[a] The average growing cycle per planting is calculated with a weighted average of all plantings, based on area. The total harvest time may be longer than the average planting time if plantings are staggered.

[b] See Table 40 for LCIs for polytunnel infrastructures.

[c] Produced crop refers to the total crop that was grown in the field, based on farmer estimates or measurements. Harvest is the amount of crop that was harvested from the field. Harvest waste percent is then calculated as the total weight of crop wasted during harvest (i.e., sorted out in the field) divided by the total produced crop. All harvest waste is assumed to be returned to the soil unless otherwise denoted.

[d] This harvest waste is composted.

[e] This refers to harvested rainwater used for irrigation. This is included for informational purposes only; harvested rainwater is not counted as water consumption within the utilised impact assessment method.

[f] The shed area used to house cultivation equipment has been allocated to kale based on time of use and / or weight or area allocation, depending on which is more appropriate. Note that the amount of shed space modelled here is only that specified by the farmer; when the farmer was unable to provide an estimate of this, shed space has been modelled using ecoinvent databases, listed as 'ecoinvent'.

Table 14 *– Fertility and pesticide inputs for UK kale cultivation*

Inputs per gross m^2 cultivated area	**Unit**	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-1**	**R-C-2**	**R-C-3**	**R-C-4**	**R-C-5**
Cover crop area, allocated to kale	$m^2a\ m^{-2}$	0.500	0.435	0.153	0	0.667	0	0.750	0	0.250	0.500	0.089	0
Cover crop type (% by seed weight in mix)	type	Rye (70%), vetch (30%)	Rye (65%), clover (35%)	Clover (100%)	n/a	Rye (65%), clover (35%)	n/a	Vetch (70%), rye (30%)	n/a	Radish (62%), mustard (38%)	Rye (75%), vetch (15%), clover (10%)	Phacelia (100%)	n/a
Cover crop seed per area of cover crop, allocated to kale	$kg\ m^{-2}$	0.025	0.0004	0.0003	0	0.0008	0	0.018	0	0.0006	0.005	0.0002	0
Compost, purchased	$kg\ m^{-2}$	0	6.52	0	0	0	3.47	0	0	0	0	0	0
Compost, homemade	$kg\ m^{-2}$	0	0	0	19.6	0	0	0	0	0	0	0	0
Fresh manure	$kg\ m^{-2}$	0	0	0	0	0.333	3.30	0	0	0	0	0	0
Composted manure	$kg\ m^{-2}$	0	0	1.32	0	0	0	0	0	0	0	0	0
Dried, pelleted manure	$kg\ m^{-2}$	0	0.082	0	0	0	0	0	0	0	0	0	0
Lime, as $CaCO_3$	$kg\ m^{-2}$	0	0	0	0	0	0	0	0.036	0	0.133	0	0
Other organic fertiliser mixes	$kg\ m^{-2}$	0	0	0	0	0	0	0	2.6E-04	3.4E-04	0	1.6E-04	6.0E-06
Mineral NPK fertilisers	$kg\ m^{-2}$	0	0	0	0	0	0	0	0.044	0.092	0.162	0.128	0.121
Mineral micronutrient fertilisers	$kg\ m^{-2}$	0	0	0	0	0	0	0	0	3.1E-04	0	5.9E-03	8.0E-04
Total N, from cover crop residues	$g\ m^{-2}$	15.3	14.2	4.30	0	18.9	0	20.6	0	2.03	18.8	1.28	0
Total N, from fertilisers	$g\ m^{-2}$	0	53.5	9.27	168	1.37	47.9	0	6.72	13.9	26.6	17.1	17.9
Soluble N, from fertilisers	$g\ m^{-2}$	0	3.11	0.464	7.56	0.273	3.26	0	6.73	13.9	26.6	17.1	17.9
P_2O_5, from fertilisers	$g\ m^{-2}$	0	24.1	2.65	79.4	0.5	19.7	0	4.48	0.075	2.33	6.25	5.43
K_2O, from fertilisers	$g\ m^{-2}$	0	45.6	9.27	124	1.93	37.8	0	4.49	10.8	14.5	7.57	14.8
Pesticides, active ingredients (ai)	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.126	0.481	0.4546	0.282	0.312
…Insecticides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.035	0.051	0.0428	0.022	0.019
…Herbicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.052	0.163	0.3324	0.145	0.241
…Fungicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.034	0.267	0.0723	0.082	0.049
…Molluscicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.005	0	0.0072	0.033	0.002
Compost / manure transport [a]	km	0	46	6	0	6	13	0	0	0	0	0	0
Other input transport [a]	km	98	279	213	0	405	13	138	149	259	165	205	63

[a] This is the average one-way distance that compost and other inputs travel from the place of sale / collection to the farm. This value does not include other transport, such as from manufacturing centres to shops.

Table 15 *– Materials and equipment for UK kale cultivation, in g (gross m^{-2})*

Item	Material	Process	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Netting	HDPE or PP	Extrusion, weaving	2.29 (12)[a]	13.7 (6)	12.6 (14)	0.739 (30)[b]	0	0	5.20 (30)	0	2.13 (5)	0	0	0
Plastic mulch	LDPE	Extrusion, film	0	0	0	0	0	0	0	5.22 (4.5 uses)	0	0	1.11 (2 uses)	0
Fleece	PP	Spunbond	0	0	0	0	1.9 (12)	0	0	0	0	0	0	3.7 (1)
Low tunnel	Steel	Wire drawing	0	19.2 (30)	12.7 (40)[b]	0	0	0	0	0	0	0	0	0
Tyres	Rubber	Extrusion	0	0	0	335 (20)[b]	0	0	0	0	0	0	0	0
Irrigation pipes	PE	Extrusion, pipe	0	0	0	0.001 (20)	0	0	0	0.100 (40)	0	0	0	0
Irrigation mainline	PVC	Extrusion, pipe	0	0	0	0	0	0.488 (40)	0.001 (40)	0	0	0	0	0
Pond liner	Butyl	Thermoform	0	0	0	0	0	0.943 (15)	0	0	0	0	0	0
Rainwater tanks	HDPE	Injection moulded	0	0.004 (30)	0	2.55 (20)	0	0	0	0	0	0	0	0
Rainwater tanks	Steel	Section bar rolling	0	0	0	6.69 (20)	0	3.97 (15)	0	0	0	0	0	0
Crates	PP or HDPE	Injection moulded	0	0	0	1.51 (10)	0.144 (3)	0.188 (35)	0.112 (10)	0	0.179 (5)	0.291 (8)	0.138 (8)	0.465 (3)
Crates	Wood	Sawnwood	0	0	0	0	0	0	0	0.961 (5)	0	0	0	0
Shipping containers	Steel	Metal working	4.38 (40)	0	7.95 (40)	0	2.78 (40)	0	0	0	0	0	0	0
Tractor, 2-wheel [c]	Mostly steel	Varies	0	0	0.674 (38)	0	0	1.37 (40)	0	0	0	0	0	0
Tractors, (<50 HP) [c]	Mostly steel	Varies	4.41 (55)	3.27 (25)	0	0	1.36 (9000 hr)	0	1.05 (9000 hr)	0.974 (10,000 hr)	0	0	0	0
Tractors, (50-100 HP) [c]	Mostly steel	Varies	0	0	0	0	0	0	0.82 (5100 hr)	0.30 (10,000 hr)	0	0.043 (9000 hr)	0.423 (10,000 hr)	0
Tractors, (>100 HP) [c]	Mostly steel	Varies	0.26 (9,000 hr)	0	0	0	1.09 (9000 hr)	0	0	0.399 (10,000 hr)	5.16 (9000 hr)	0.275 (9000 hr)	0.374 (10,000 hr)	2.0 (9000 hr)
Implements [c]	Mostly steel	Varies	8.34 (varies)	0.90 (20-30)	0.612 (775 hr)	0	4.97 (varies)	0.571 (40)	1.94 (varies)	4.95 (varies)	4.29 (varies)	0.645 (varies)	0.654 (varies)	0.293 (varies)

[a] All material amounts are provided in g (gross m^{-2}) and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated.
[b] These materials have been reclaimed, and thus only transport and end of life waste burdens are attributed to these materials; no production burdens are considered.
[c] All tractors, implements, and other cultivation equipment are modelled using ecoinvent; amounts have been allocated to kale based on the time of use for the crop, out of the total equipment lifetime, in hours.

2.1.7.3.3 Processing and storage

Following the harvest of kale, the crop may be washed, chopped, and then packaged or bunched, depending on the farm. The crop is generally stored in cold storage before and after packaging, before final transport. Thus, the LCIs from processing and storage on the farm (or relevant farm packhouses) are considered within this section, listed per kg sellable crop. These LCIs do not include transport to the packhouse (and associated fuel use), as this has already been modelled within the cultivation phase. Further, any additional crop storage during distribution (such as at regional distribution centres) is not considered here, as it is included within the distribution phase.

Table 16 and Table 18 provide an overview of major resource flows during kale processing and storage for U.S. and UK farms, respectively, as well as listing processing waste amounts (out of total harvest) and infrastructure use, such as packhouses and cold storage facilities. Propane use is also considered for UK conventional farms, which is used for forklifts within packhouses. Table 17 and Table 19 then provide an overview of the materials used to package and store the crop before final distribution, for U.S. and UK farms, respectively. These include packaging materials such as cardboard boxes, plastic films and bags, rubber bands for bunching, as well as materials used to transport and store the product, such as plastic crates and wooden or plastic pallets. The weights of packaging materials were either measured directly on the farm, or provided by specific suppliers. Lifetimes of materials are listed in within parentheses, in years unless otherwise stated; in some cases, lifetime is listed based on number of uses, where one use is assumed to be one crop packaged or transported, depending on the item. Many organic farms may reclaim materials from other businesses (e.g., plastic crates from supermarkets, cardboard boxes from other farms at markets) and then use these items for their own packaging and storage needs. All reclaimed materials are designated with footnotes.

The processing and storage of kale differs between farms in the U.S. and UK. On conventional U.S. and UK farms, kale is normally washed; this is also a common practice on U.S. organic farms, but less so on UK organic farms. This provides the main source of water use, although some additional water use may be considered for misters within cold stores or for packaging kale with ice for transport, the latter of which is only a standard practice on the U.S. conventional farm. Additionally, some conventional farms in the UK chop their kale before packaging, leading to additional energy use (R-C-2, R-C-4, and R-C-5); this process is not performed on any other farms in the UK or U.S.

Kale is then generally stored in cold storage facilities temporarily before being distributed. This is the case on all conventional farms in both countries, and most organic U.S. farms; however, less UK organic farms require cold storage for their kale crops, due to the generally cooler temperatures in the UK and the fact that crops are generally sold soon after harvest. Many U.S. organic farms utilise A/C units to create low-cost cool rooms or walk-in coolers using CoolBot systems (https://www.storeitcold.com/), as described in Section 2.1.7.2.8. These cool rooms are usually kept at temperatures of around 4-10°C (40-50°F) for kale. However, other farms in both the U.S. and UK generally use refrigerated cold storage facilities, either as walk-in coolers within a packhouse, or, in some cases, in refrigerated shipping containers. Finally, some small farms in the U.S. utilised standing refrigerators for crop storage. Further details on the modelling of cold storage infrastructure are provided in Section 2.1.7.5.2.

Packaging methods also vary across U.S. and UK farms. In the U.S., kale is normally bunched using rubber bands or twist ties. These may then be transported loose in cardboard boxes or plastic crates. For U.S. conventional farms, there is some uncertainty with whether additional packaging is applied from other retail chains (e.g., if selling through wholesalers). However, for the main supermarket chains considered in this LCA (as specified by the farmer), kale is usually sold unpackaged, as bunches in plastic crates under a mister. Thus, it can be assumed that there is unlikely to be additional packaging.

On the other hand, UK farms generally do not bunch kale, but instead package portions individually, usually in plastic bags or packs. The UK supermarket chains considered in this LCA generally do not use cool misters as done in the U.S., and thus all kale is packaged rather than being sold in loose bunches. The UK conventional farms that sell through supermarket chains perform all packaging for the retail chain on-site (e.g., using the supermarket's specific packaging and branding). Conventional farms may also package a portion of kale loose into cardboard boxes or plastic crates with plastic liners (usually for conventional farms). For the latter case, the plastic crates used are normally part of the returnable IFCO system (https://www.ifco.com/), where crates are returned by supermarkets to IFCO depots, washed, and redistributed to farms for continued use. The washing of crates has not been modelled due to limited information on this process; however, the lifetime of these crates has been assumed as 75 uses, as per the supplier. Finally, for conventional farms in both countries, wooden or plastic pallets ('dolavs') are also used to stack crates or boxes for storage and transport. The lifetime of wooden pallets is estimated as thirty uses based on Carrano *et al*. (2014), where one use is considered to be one transport journey.

For organic farms in both the U.S. and UK, there are some cases where the crop is stored and transported in plastic crates or cardboard boxes that are reused over the season or over several years; in these cases, there is no additional packaging considered. However, some organic farms may also require additional packaging when bags or boxes are used for multiple food items (e.g., for veg boxes and CSAs). In this case, packaging has been allocated to kale based on the weight of kale out of the total weight of other crops in the bag or box.

Table 16 – U.S. kale processing and storage: major resource flows

Flows	**Unit**	**U-O-1**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-O-3**	**R-C-3**
Sellable crop, total	kg	143	136	81	69	181	536	544	311	3,401,943
Sellable crop, per gross m^2	kg m^{-2}	1.42	1.95	0.72	0.75	1.63	0.72	1.04	0.70	2.10
Processing waste, % of harvest [a]	%	1.5%[b]	0%	4.5% [b]	0%	10.1% [b]	0%	0%	2.0% [c]	4.0% [d]
Resources (kg sellable crop)$^{-1}$										
Energy use, GA mix	kWh	0	0	0.506	0	0.573	0.440	0.715	0.435	0.069
Energy use, solar	kWh	1.33	0	0	0	0	0	0	0.405	0
Water use	L	31.5	0	90.5	66.5	9.39	12.0	4.69	25.4	17.2
Packhouse area, allocated [e]	m^2a	0.005	0.005	0.046	0	0	6.94E-04	4.10E-03	3.22E-03	2.34E-05
Cold storage area, allocated [e]	m^2a	0	0	0	0	1.64E-03 [f]	1.55E-03 [f]	1.60E-03	1.59E-03[f]	1.10E-05
Fridge / freezer volume [g]	m^3a	0.072	0	0.0007	0	0	0	0	0	0

[a] Processing waste is listed as the percent of crop wasted during processing and storage, out of the total harvested crop.
[b] This indicates processing waste that is composted; [c] fed to animals on the farm; and [d] donated to local churches or charities.
[e] Packhouses may be steel or wooden shed structures, whilst cold storage areas are generally insulated cement or steel rooms that may or may not employ refrigeration. Space is allocated to kale based either on the space used for the crop and the time used out of the year, or using weight allocation over the year (out of all crops produced on the farm), depending on which is more appropriate.
[f] This indicates cold storage areas that utilise air conditioning for cooling, rather than conventional refrigeration, by using the Coolbot cooling system (https://www.storeitcold.com/).
[g] These refer to standing refrigerators and chest freezers.

Table 17 – Materials for packaging and storage of U.S. kale, in g (kg sellable crop)$^{-1}$

Item	Material	Process	U-O-1	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-O-3	R-C-3
Cardboard boxes	Containerboard	Board box production	0 [a]	0	0	0	0	0	24.2 (1)[b]	0	54.1 (1 use)
Plastic bags / film	LDPE	Extrusion, film	0	3.15 (2.5 uses)	15.2 (1 use)	0	0	2.12 (1 use)	16.7 (1 use)	0.50 (1 use)	0
Plastic bags	HDPE	Extrusion, film	0	6.31 (1 use)	10.4 (1 use)	0	0	0	0	3.02 (1 use)	0
Plastic bags	PLA	Extrusion, film	0	0	11.9 (1 use)	0	0	0	0	0	0
Rubber bands, for bunching kale	Rubber	Extrusion	1.71 (1 use)	0	0.474 (1 use)	2.57 (1 use)	1.03 (1 use)	1.57 (1 use)	0.268 (1 use)	0	0
Twist ties, for bunching kale	60% LDPE; 40% steel	Extrusion, film; wire drawing	0	0	0	0	0	0	0	0	2.10 (1 use)
Storage crates	PP or LDPE	Injection moulded	2.79 (5)	0	3.97 (7)	0.490 (10)	1.43 (10)	0.099 (20)	0	0	0.295 (75 uses)
Crates / coolers	HDPE	Injection moulded	0	0	0.868 (30)	0	0.434 (30)	3.72 (20)	0.818 (20)	1.04 (5)	0
Pallets	Wood; steel	Sawnwood; hot rolled	0	0	0	0	0	0	0	0	1.14 (30 uses)
Shipping containers for crop storage	Steel	Metal working, welding	0	0	0	42.1 (40)	0	0	0	0	0
Average transport of packaging materials (km) [c]	n/a	n/a	13	57	29	9	9	170	225	57	223

[a] All material amounts are provided in g (kg sellable crop)$^{-1}$ and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated. If lifetime is designated in 'uses', it is assumed that one use is either 1 kg of crop packaged (in the case of packaging) or 1 kg crop transported, in the case of transport materials like pallets or crates.

[b] These boxes are reused just for kale over the entire harvest season, so this considers the total amount used over the season, per kg sellable kale.

[c] This is the average one-way transport distance of packaging materials, from the place of purchase to the farm or packhouse site, in km. This distance does not include other transport, such as from manufacturing centres to shops, which is accounted for within modelled ecoinvent processes.

Table 18 – UK kale processing and storage: major resource flows

Flows	Unit	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Sellable crop, total	kg	200	170	154	71.8	708	870	1,275	5,550	18,710	980,000	442,960	1,190,406
Sellable crop, per gross m^2	kg m^{-2}	0.26	0.92	0.51	2.15	0.79	2.07	0.51	0.80	0.89	1.11	0.47	0.60
Processing waste, % of harvest [a]	%	0	0	0	0	0	0	0	7.5% [b]	5.0% [c]	2.60% [b]	2.0% [c]	30% [c]
Resources (kg sellable crop)$^{-1}$													
Energy use, GB mix	kWh	0	0	0	0	0.102	0.000	0.008	0.071	1.42	0.057	0.133	0.465
Energy use, solar	kWh	0	0	0	0	0	0	0	0	0.356	0	0.047	0
Propane use	kg	0	0	0	0	0	0	0	0.005	0	0.005	0.007	0.003
Water use	L	0	0	0	0	0	0	0	0	1.55	0.057	0.257	4.03
Packhouse area, allocated [d]	m^2a	0.0011	0	0	0.0155	0.0035	0.0328	0.0115	0.0017	0.0018	0.0004	0.0002	0.0003
Insulated storage area, allocated [d]	m^2a	0	0	0.0017[e]	0	0	0	0	0	0	0	0	0
Refrigerated cold storage area, allocated [d]	m^2a	0	0	0	0	1.24E-03[e]	0	1.21E-05	5.58E-05	1.53E-04	1.18E-04	3.99E-04	9.61E-04

[a] Processing waste is listed as the percent of crop wasted during processing and storage, out of the total harvested crop.
[b] This indicates processing waste that is turned back into the field soil, or [c] fed to animals on the same farm or other farms. The latter case takes into account any relevant transport burdens.
[d] Packhouses may be steel or wooden shed structures, whilst cold storage areas are generally insulated cement or steel rooms that may or may not employ refrigeration. Space is allocated to kale based either on the space used for the crop and the time used out of the year, or using weight allocation over the year (out of all crops produced on the farm), depending on which is more appropriate.
[e] These both refer to shipping containers that are used for crop storage; for PU-O-2, this is just an insulated shipping container, but for PU-O-4, the shipping container includes a refrigeration system. Material use for the shipping container structures are also included in Table 19.

Table 19 – *Materials for packaging and storage of UK kale, in g (kg sellable crop)$^{-1}$*

Item	Material	Process	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Cardboard boxes	Container-board	Board box production	0 [a]	7.75 (6 uses)	12.6 [b] (6 uses)	0	0	0	0	185 (1 use)	48.5 (1 use)	130 (1 use)	0	0
Plastic bags / film	PP	Extrusion, film	0	0	0	0	0	0	0	0	5.49 (1 use)	11.0 (1 use)	65.8 (1 use)	824 (1 use)
Plastic bags / film	LDPE	Extrusion, film	17.3 (1.5 uses)	5.91 (2.5 uses)	5.97 (1 use)	0	20.8 (1 use)	9.96 (1 use)	1.18 (1 use)	28.5 (1 use)	4.19 (1 use)	0.042 (1 use)	0	0
Plastic bags	HDPE	Extrusion, film	0	0	0	0	0	0	0	0	0	0	0	0
Biodegradable plastic bags	Biodegradable LDPE (d2w)	Extrusion, film	0.082 (2)	0	0	2.09 (1 use)	2.80 (2.5 uses)	0	0	0	0	0	0	0
Reusable tote bags	Jute	Textile production	0	0	0	0.359 (23)	0	0	0	0	0	0	0	0
Storage crates	PP / LDPE	Injection moulded	0	0	0	0.705 (10)	0.345 (25)[b]	0.091 (35)	0.387 (10)	1.45 [c] (75 uses)	1.22 [c] (5)	0	10.3 [c] (75 uses)	0
Plastic box pallets (dolav)	HDPE	Injection moulded	0	0	0	0	0	0	0	0	0	0.261 (7.5)	0	2.24 (2.5)
Crates	Wood	Sawnwood	0	0	0	0	0	0	0	1.83 (9)	0	0	0	0
Pallets	Wood; steel	Sawnwood; hot rolled	0	0	0	0	0	0	0	5.02 (30 uses)	0	3.85 (30 uses)	6.98 (30 uses)	0
Shipping containers for crop storage	Steel	Metal working, welding	0	0	11.0 (40)	0	2.74 (40)	0	0	0	0	0	0	0
Average transport of packaging materials (km) [d]	n/a	n/a	49	23	24	122	185	301	217	71	93	114	481	114

[a] All material amounts are provided in g (kg sellable crop)$^{-1}$, and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated. If lifetime is designated in 'uses', it is assumed that one use is either 1 kg of crop packaged (in the case of packaging) or 1 kg crop transported, in the case of transport materials like pallets or crates.

[b] These materials have been reclaimed, and thus only transport and end of life waste burdens are attributed to these materials; no production burdens are considered.

[c] These are IFCO plastic crates (https://www.ifco.com/), which are part of a nation-wide reusable packaging container system where crates are returned from retail chains, washed, and redistributed for use. The estimated lifetime is 75 uses as per the supplier, unless otherwise specified by the farmer.

[d] This is the average one-way transport distance of packaging materials, from the place of purchase to the farm or packhouse site, in km. This distance does not include other transport, such as from manufacturing centres to shops, which is accounted for within modelled ecoinvent processes.

2.1.7.3.4 Distribution

The distribution phase considers the transport of the crop to its final point of sale, as well as any additional storage burdens during transport. Final distribution is the last stage of the lifecycle considered for the crop; any additional burdens that occur after the crop arrives at its final point of sale (e.g., burdens from the retail environment) are not considered.

Table 20 and Table 21 provide an overview of the kale transport distances and resource use for crop storage in regional distribution centres (RDCs), per kg sellable crop, for U.S. and UK farms, respectively. All sellable crop is assumed to be transported, so sellable crop is equivalent to the transported crop. Although values for total sellable crop on each farm were originally provided within processing LCIs (Table 16 and Table 18), they have been reproduced here for ease of reference. No waste is considered during transport, due to limited information.

The average one-way trip distance is provided for each farm in km, which is the average distance that one kg of crop travels to its final point of sale. It is calculated as a weighted average of all trip distances travelled to different points of sale, with the types of sale points used by each farm being previously designated in Section 2.1.7.1 (Table 1 and Table 2). Organic farms all sell directly to their final consumer (e.g., at farmer's markets, through local shops or restaurants, or through CSA and veg box schemes), and these points of sale are always located within the same state for U.S. farms or region for UK farms. For farms that sell all crop on the farm (US-PU-O-1 and US-PU-O-3), transport distances are assumed to be zero; for other farms that sell only a portion of their crop on the farm, this is taken into account within the weighted average one-way distance.

Generally, it is seen that crops travel farther distances to their final points of sale from rural farms, which is expected as these crops are being transported into cities for sale. Some UK organic farms (PU-O-1, PU-O-2, PU-O-4) also sell veg boxes through city-wide delivery routes (either for home delivery or to pick-up points), which contribute to additional transport burdens. Still, the average one-way transport distances for all organic farms are lower than corresponding conventional farms in the same country, which transport crops mainly to retail chains. In this case, crops first travel to an RDC and then are transported to a variety of shops that may be located across the country. In particular, the U.S. conventional farm sells kale throughout the East coast of the U.S., while several UK conventional farms sell through retail chains or food recipe boxes that are distributed throughout Great Britain. Transport distances for U.S. farms tend to be higher, due to the greater country scale.

Modes of transport also differ between organic and conventional farms, which is why different allocated transport distances per kg sellable crop are provided in Table 20 and Table 21. All organic U.S. farms transport kale via large size passenger cars, such as pick-up trucks, mini-vans, or transit vans, using petrol as the fuel input. As previously explained in Section 2.1.7.2.8, this is modelled using ecoinvent v.3.7.1 passenger car processes, which are listed in units of km. Thus, the total round-trip distances travelled to all sale points over the harvest season must be calculated and then allocated to the kale crop. Several UK organic farms also transport crops via either petrol or diesel passenger cars, with one (PU-O-2) using electric bicycles to transport a portion of their crop through a home delivery route.

On the other hand, all conventional farms transport kale either using lorries / semi-trucks (16-32 metric tonnes) or diesel transit vans (3.5-7.5 metric tonnes). In some cases, this transport occurs through refrigerated lorries, which is designated through footnotes in the tables (for UK farms only). Several UK organic farms also use diesel transit vans for transport, especially those transporting veg boxes through city-wide delivery routes (PU-O-1, PU-O-2, and PU-O-4). Transport through lorries and diesel vans are modelled within ecoinvent in terms of kg*km, and thus the average one-way distance for these modes of transport has been listed in the tables using this unit. Finally, for conventional farms only, the average distance travelled to each RDC, and from each RDC to shops, has been listed. This transport may occur either through lorries or diesel transit vans, with transport distances based on estimates as designated in Section 2.1.7.2.8.

Finally, additional storage burdens are considered for crops that travel through RDCs which, in this case, are only crops sold by conventional farms to retail chains or food recipe boxes. Burdens associated with crop storage in RDCs include operational electricity use; cold storage electricity use; heating via natural gas; and water use. These have been estimated based on PEFCR guidance for food product storage in distribution centres (European Commission, 2018), with more detail provided in Section 2.1.7.2.9 for resource flows and Section 2.1.7.5.2 for RDC infrastructure and storage space calculations. All kale is assumed to be stored in cold storage facilities.

Table 20 – U.S. kale distribution: major resource flows

Flows	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-O-3	R-C-3
Transported / sellable crop, total	kg	143	136	81	69	181	536	544	311	3,401,943
Average one-way trip distance [a]	km	12	0	33	0	22	95	69	12	1,280
Flows (kg sellable crop)$^{-1}$										
Transport via large passenger car, allocated [b]	km	0.12	0	11.74	0	0.11	2.35	1.43	0.74	0
Transport via lorry [c]	kg*km	0	0	0	0	0	0	0	0	1,280
…transport from farm to RDC	kg*km	0	0	0	0	0	0	0	0	1,070
…transport from RDC to shops	kg*km	0	0	0	0	0	0	0	0	210
RDC storage area (cold storage) [d]	m^3a	0	0	0	0	0	0	0	0	2.01E-04
Electricity use, RDC [d]	kWh	0	0	0	0	0	0	0	0	8.09E-03
Heat use (natural gas), RDC [d]	MJ	0	0	0	0	0	0	0	0	7.23E-04
Water use, RDC [d]	L	0	0	0	0	0	0	0	0	2.45E-05

[a] This is the average trip distance travelled by the kale crop to all points of sale. It is calculated as a weighted average of all trip distances travelled. For organic farms, this is weighted based on the total number of trips; for conventional farms, this is weighted based on the proportion of crop sold at each sale point. It also considers crop that is sold on the farm as a '0 km' trip distance.
[b] Transport via passenger cars is modelled in ecoinvent in terms of km. Thus, the total round-trip distance travelled by all kale crop (summing each journey) is calculated and then allocated to kale based on its proportion out of all crops transported. This is the figure represented in this row.
[c] Transport via lorry is modelled in ecoinvent as kg*km. Thus, kg*km per kg sellable crop is the same as the average trip distance provided in the second row.
[d] This is calculated based on the volume of the packaged crop (e.g., box volume), * 3 for chilled crops, and assuming a storage of 1 week as per PEFCR guidance (European Commission, 2018). This value considers both the infrastructure for the total RDC centre, as well as specific infrastructure for cold storage. Electricity, heat, and water use is also based on PEFCR guidance.

Table 21 *– UK kale distribution: major resource flows*

Flows	Unit	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-1	R-C-2	R-C-3	R-C-4	R-C-5
Transported / sellable crop, total	kg	200	170	154	71.8	708	870	1,275	5,550	18,710	980,000	442,960	1,190,406
Average one-way trip distance [a]	km	19	28	155	6	150	74	33	176	202	270	456	313
Flows (kg sellable crop)$^{-1}$													
Transport via large passenger car, allocated [b]	km	0.118	0.101	0.328	0.138	0	0	0.153	0	0	0	0	0
Transport via electric bike, allocated	km	0	0	0.006	0	0	0	0	0	0	0	0	0
Transport via diesel transit van	kg*km	0	3.72	115	0	150	74	0	133	86	0	0	0
Transport via lorry [c]	kg*km	0	0	0	0	0	0	0	44	116	270	456 [d]	313 [d]
…transport from farm to RDC	kg*km	0	0	0	0	0	0	0	64	86	243	364	82
…transport from RDC to shops	kg*km	0	0	0	0	0	0	0	99	116	26	92	232
RDC storage area (cold storage) [e]	m^3a	0	0	0	0	0	0	0	6.45E-04	1.12E-03	4.61E-04	1.32E-03	7.88E-04
Electricity use, RDC [e]	kWh	0	0	0	0	0	0	0	2.60E-02	4.52E-02	1.86E-02	5.31E-02	3.18E-02
Heat use (natural gas), RDC [e]	MJ	0	0	0	0	0	0	0	2.32E-03	4.04E-03	1.66E-03	4.75E-03	2.84E-03
Water use, RDC [e]	L	0	0	0	0	0	0	0	7.87E-05	1.37E-04	5.63E-05	1.61E-04	9.61E-05

[a] This is the average trip distance travelled by the kale crop to all points of sale. It is calculated as a weighted average of all trip distances travelled. For organic farms, this is weighted based on the total number of trips; for conventional farms, this is weighted based on the proportion of crop sold at each sale point. It also considers crop that is sold on the farm as a '0 km' trip distance.
[b] Transport via passenger cars is modelled in ecoinvent in terms of km. Thus, the total round-trip distance travelled by all kale crop (summing each journey) is calculated and then allocated to kale based on its proportion out of all crops transported. This is the figure represented in this row.
[c] Transport via lorry is modelled in ecoinvent as kg*km. Thus, kg*km per kg sellable crop is the same as the average trip distance provided in the second row.
[d] This transport occurs through refrigerated lorries.
[e] This is calculated based on the volume of the packaged crop (e.g., box volume), * 3 for chilled crops, and assuming a storage of 1 week as per PEFCR guidance (European Commission, 2018). This value considers both the infrastructure for the total RDC centre, as well as specific infrastructure for cold storage. Electricity, heat, and water use is also based on PEFCR guidance.

2.1.7.4 Tomato LCIs

As for kale production, this section provides an overview of the resources and material flows modelled for tomato lifecycles on case study farms in the U.S. and UK. This includes data from nine farms in the U.S., including one urban organic farm (U-O), three peri-urban organic farms (PU-O), three rural organic farms (R-O), and two rural conventional farms (R-C). In the UK, nine case study farms are also considered, which include two urban organic farms, four peri-urban organic farms, two rural organic farms, and one rural conventional farm; however, for conventional production, two cases are explored based on the energy source used within the glasshouses, which include conventional electricity through the national grid and heating through natural gas (R-C-NG), as well as energy from combined heat and power (R-C-CHP). These different energy sources are only considered for the energy supply for glasshouses (i.e., during seedling production and cultivation); for all other energy use, such as in packhouses, electricity from the national grid is assumed for both cases. In the following sections, lifecycle inventories are listed separately for seedling production; crop cultivation; on-farm processing, packaging, and storage; and distribution to final points of sale.

2.1.7.4.1 Seedlings

Lifecycle inventories for tomato seedlings are provided in Table 22 through Table 24 for U.S. tomato farms and Table 25 through Table 27 for UK tomato farms. These inventories are provided in a similar fashion to those for kale seedlings. In the first of these sets of tables, an overview of resource flows is provided, specifying the number of seedlings germinated, where this takes place (e.g., on the farm or at a nursery), the infrastructure used, and energy and water use. The second provides an overview of the growing media or potting mix components used and any fertilisers or pesticides applied, including organic fertilisers. Finally, the third table lists all non-infrastructural material uses. These materials have been modelled using ecoinvent v.3.7.1, with the specific manufacturing processes used also provided in these tables. Material lifetimes as designated by the farmer or supplier are also included, listed in parentheses next to amounts used. All material amounts listed have already been allocated based on the time of use for the crop lifecycle and the lifetime. As for the kale seedling inventories, all resources and materials are listed per seedling transplanted, unless otherwise specified.

Similar to kale seedling production, the U.S. organic farms tend to use a wider range of growing media and organic fertilisers during seedling production than UK organic farms. Again, all U.S. farms germinate seedlings in polytunnels or low tunnels, while some farms in the UK, especially the larger-scale nurseries, tend to use glasshouses for production. Heating is used more commonly for tomato germination than seen for kale germination, with many organic farms using heat mats or heating coils buried in sand on benches to heat growing trays individually. The germination of seedlings for UK conventional tomato production occurs outside of the country, in the Netherlands. Further details on the LCI for UK conventional tomato seedlings is provided in Appendix A, which lists all inputs modelled and also provides the data sources used.

Table 22 – *Resource flows for U.S. tomato seedlings*

Flows	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Seedlings germinated	Number	220	648	684	1,310	1,008	2,432	513	271,765	1,366,200
Seedlings planted	Number	176	263	670	873	600	2,400	410	231,000	1,188,000
Total germination time	Weeks	16	5.56	5.52	7	8	24	21	14	20.25
Germination time per round	Weeks	8	5.56	5.52	7	8	8	7	7	5.68
Polytunnel / glasshouse type [a]	Type	US-P-1	US-P-2	US-P-2	Low tunnel	US-P-1	US-P-1	US-P-1	US-P-3	US-P-2
Germination location	Type	On farm	Two nurseries	Two nurseries	On farm	On farm	On farm	On farm	On farm	On farm (75%); nursery (25%)
Transport from nursery	km, one way	0	337	103	0	0	0	0	0	504
Resources (seedling transplanted)$^{-1}$										
Area use [b]	m^2	0.10	0.01	0.001	0.003	0.07	0.005	0.018	0.003	0.0014
Energy use, GA mix	kWh	0.90	0.004	0.030	0	0.301	0.030	1.40	0.003	0.014
Energy use, solar	kWh	0	0	0	0	0	0	1.31	0	0
Propane use	kg	0	0.005	0.015	0	0	0	0	0	0.018
Water use	L	13.4	0.270	0.350	0.17	4.65	5.44	0.678	1.00	0.290

[a] See Table 40 for LCIs for polytunnel and glasshouse infrastructures.
[b] This is the area used over the total germination time, per seedling transplanted

Table 23 *– Fertiliser, growing media, and pesticide inputs for U.S. tomato seedlings, in g (seedling transplanted)$^{-1}$*

Inputs	**U-O-1**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**R-O-1**	**R-O-2**	**R-O-3**	**R-C-1**	**R-C-2**
Sand	0	0	0	2.06	0	0	2.58	0	0
Peat moss	79.4	22.9	4.40	1.09	83.9	2.60	16.3	2.51	7.62
Perlite / Vermiculite	40.1	3.64	0.85	0.51	69.0	0	5.13	0.40	1.29
Wood chips	0	0	0	35.0	0	0	0	0	0
Lime	0.35	0.66	0.12	0	1.67	0	0	0.052	0.40
Compost, purchased	318	0	0	0	1,323	18.2	0	0	0
Worm castings	0	1.36	0.27	0	0	0	22.6	0	0
Azomite	0.23	0	0	0	3.46	0	0	0	0
Blood meal / feathermeal	0.23	0	0	0	12.1	0	0	0	0
Kelp meal / fish emulsion	0.12	0.82	2.06	0	0	0	2.49	0	0
Alfalfa meal	0.47	0	0	0	0	0	0	0	0
Other organic fertiliser mixes	0	0	0	4.63	0	0	0	0	0
Mineral NPK fertilisers	0	7.81	0	0	0	0	0	0.157	0.025
N, in all fertilisers	3.25	1.62	0.050	0.014	15.0	0.184	0.097	0.016	0.003
P_2O_5, in all fertilisers	0.80	1.60	0.081	0.0046	3.33	0.455	0.075	0.031	0.012
K_2O, in all fertilisers	0.93	1.59	0.025	0.005	3.87	0.053	0.025	0.031	0.002
Pesticides (g ai) [a]	0	0	0	0	0	0	0	0	0
Compost transport (km) [b]	282	0	0	0	214	813	0	0	0
Other input transport (km) [b]	14.0	333	139	8.69	295	813	182	1,358	973

[a] These are the pesticides applied, in grams active ingredient (ai) per seedling transplanted.

[b] This is the average one-way distance that compost and other inputs travel from the place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops.

Table 24 – Material use for U.S. tomato seedlings, in g (transplanted seedling)$^{-1}$

Items	Material	Process	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Pots and trays	PS	Injection moulded	1.66 (3.42) [a]	4.54 (1)	0.271 (1)	0.489 (3)	0.151 (7)	0	0.510 (3.15)	0	0
Trays	PS	Foaming	0	0	0	0	0	0	0	0.064 (10)	0.101 (4.4)
Long-life trays	Acrylic	Injection moulded	0.155 (30)	0	0	0	0	0.122 (30)	0	0	0
Humidity domes	PET	Injection / blow moulded	0	0	0	0	0	0	0.184 (3.5)	0	0
Low tunnel	PE	Extrusion, film	0	0	0	0.044 (6)	0	0	0	0	0
Low tunnel	PVC	Extrusion, pipe	0	0	0	0.006 (40)	0	0	0	0	0
Heat mats, cover	Rubber	Extrusion	6.32 (5)	0	0.065 (5)	0	0.223 (12)	0	0	0	0
Heat coil	Copper	Wire drawing	0.375 (5)	0	0.004 (5)	0	0.013 (12)	0.008 (8)	0	0	0
Material transport to farm (km) [b]	n/a	n/a	724	150	252	235	1,364	262	901	80	284

[a] Material amounts are provided in g (transplanted seedling)$^{-1}$, allocated based on time of use and material lifetime. Material lifetimes, in years, are provided in parentheses (), but note that these have already been applied to the provided material amounts.
[b] This provides the average one-way distance that materials are transported from their place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops. However, these distances are accounted for within modelled ecoinvent processes.

__Table 25__ – Resource flows for UK tomato seedlings

Flows	**Unit**	**U-O-1**	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-NG R-C-CHP**
Seedlings germinated	Number	180	900	120	300	100	250	360	330	550,000
Seedlings planted	Number	180	650	100	160	100	250	360	300	500,000
Total germination time	Weeks	16	11	3	9	9	7.5	4	4	12
Germination time per round	Weeks	16	11	3	9	9	7.5	4	4	12
Polytunnel / glasshouse type [a]	Type	UK-P-1	UK-P-1	UK-G-1	UK-G-1	Glasshouse + reclaimed tunnel	UK-P-2	Glasshouse (ecoinvent)	Glasshouse (ecoinvent)	Glasshouse (ecoinvent)
Germination location	Type	On farm	On farm	On farm	On farm	Nursery + farm	On farm	Nursery	Nursery	Nursery
Transport from nursery	km, one way	0	0	0	0	4	0	330	290	618
Resources (seedling transplanted)$^{-1}$										
Area use [b]	m^2	0.67	0.49	0.074	0.0061	0.018	0.043	0.033	0.037	0.44
Electricity use [c]	kWh	0.115	0.017	0.235	0.064	0	0.247	0.007	0.007	0.315
Heat use [c]	kWh	0	0	0	0	0	0	0	0	2.05
Natural gas use, for heat	m^3	0	0	0	0	0	0	0	0	0.188, R-C-NG 0.374, R-C-CHP
Wood chips, for heat	kg	0	0	0	0	0	0	0.20	0.22	0
Surplus electricity (CHP)	kWh	0	0	0	0	0	0	0	0	1.42 [d]
Water use	L	2.49	6.94	1.5	0	0.36	0.02	0.17	0.18	2.37
Water use, rainwater [e]	L	0	0	0	1.72	1.24	0	0	0	0

[a] See Table 40 for LCIs for polytunnel and glasshouse infrastructures, listed for labelled infrastructures. Large-scale glasshouses are modelled using AGRIBALYSE and not reproduced to maintain copyright principles.
[b] This is the area used over the total germination time, per seedling transplanted.
[c] For the case of R-C-CHP, the energy mix considered is combined heat and power; for all other cases it will be the electricity mix in the country of production.
[d] Surplus electricity is produced from combined heat and power generation, and this is considered only for the case of R-C-CHP. Allocation depends on the Scenario; see: Section 2.1.7.4.3.
[e] Water use from captured rainwater is listed here for information purposes, but note that this does not contribute to water consumption impacts in the impact assessment.

Table 26 – Fertiliser, growing media, and pesticide inputs for UK tomato seedlings, in g (seedling transplanted)$^{-1}$

Inputs	U-O-1	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-NG R-C-CHP
Sand	0	0	0	69.5	0	0	0	0	0
Peat moss	0	0	0	0	107	0	122	134	0
Perlite / Vermiculite	0	0	0	0	10.5	0	0	0	0
Coconut coir	12.0	30.7	10.4	0	0	35.5	0	0	103
Compost, purchased	87.0	222	10.9	0	0	257	40.5	44.6	0
Compost, homemade	275	0	0	0	0	0	0	0	0
Composted manure	0	0	0	109	0	0	0	0	0
Leaf mould	0	0	0	116	0	0	0	0	0
Other organic fertiliser mixes	0	0	0	0	0	0	2.04	2.25	0
Mineral NPK fertilisers	0	0	0	0	0	0	0	0	0.69
N, in all fertilisers	4.71	0.09	0.03	0.77	0.07	0.10	0.27	0.30	0.10
P_2O_5, in all fertilisers	1.85	0.05	0.02	0.22	0.08	0.06	0.13	0.14	0.02
K_2O, in all fertilisers	3.45	0.32	0.05	0.77	0.09	0.37	0.31	0.34	0.12
Pesticides (g ai) [a]	0	0	0	0	0	0	0	0	0.002
Compost transport (km) [b]	86.6	86.6	718	0	0	168	0	0	0
Other input transport (km) [b]	86.6	86.6	718	8.67	217	168	394	394	155

[a] These are the pesticides applied, in grams active ingredient (ai) per seedling transplanted.
[b] This is the average one-way distance that compost and other inputs travel from the place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops.

Table 27 – *Material use for UK tomato seedlings, in g (transplanted seedling)$^{-1}$*

Items	**Material**	**Process**	**U-O-1**	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-NG R-C-CHP**
Pots and trays	PS	Injection moulded	1.18 (8) [a]	0.07 (16)	0	0.29 (20)	0.12 (10)	0	0	0	0
Insulative material	PS	Foaming	0	0	0	0.002 (25)	0	0	0	0	0
Pots and trays	PP	Injection moulded	0	0	0.057 (15)	0	0.15 (10)	1.17 (9)	0.61 (5)	0.73 (5)	0.85 (75 uses)
Fleece; mesh for grow cubes	PP	Extrusion, film	0	0	0.0015 (7.5)	0.002 (7.5)	0	0	0	0	1.96 (1 use)
Humidity domes; heat mat	PET	Injection/blow moulded	0	0	0	0	0	0.074 (5)	0	0	0
Plastic liners; heat mat insulation	PE	Extrusion, film	0.54 (12)	0.35 (12)	0	0.0008 (12)	0.011 (17.5)	0	0	0	0
Drip irrigation	PE	Injection moulded	0	0	0	0	0	0	0	0	0.88 (5)
Germination chamber cladding	PC	Extrusion / thermoforming	0	0	0	0.022 (25)	0	0	0	0	0
Hoops; heat mats	PVC	Extrusion, pipe	0.25 (40)	0.16 (40)	0	0.0016 (40)	0	0	0	0	0
Heat mat	PVC	Extrusion, film	0	0	0	0	0	0.29 (3)	0	0	0
Germination chamber	Al	Section bar extrusion	0	0	0	0.085 (25)	0	0	0	0	0
Benches; chambers	Wood	Sawnwood	6.53 (20)	4.20 (20)	0	0.042 (20)	0	0	0	0	0
Heat coil	Copper	Wire drawing	0.007 (30)	0.001 (30)	0.035 (30)	0.0007 (25)	0	0	0	0	0
Heat coil	Ni-Cr alloy	Wire drawing	0	0	0	0	0	0.029 (3)	0	0	0
Heat bed	Sand	n/a	34.4 (30)	22.1 (30)	0	0	0	0	0	0	0
Material transport to farm (km) [b]	n/a	n/a	77	92	126	55	75	103	75	75	75

[a] Material amounts are provided in g (transplanted seedling)$^{-1}$, allocated based on time of use and material lifetime. Material lifetimes are provided in parentheses () and listed in years unless otherwise specified; note that these have already been applied to the provided material amounts.
[b] This provides the average one-way distance that materials are transported from their place of sale to the farm. This distance does not include other transport, such as from manufacturing centres to shops. However, these distances are accounted for within modelled ecoinvent processes.

2.1.7.4.2 Cultivation

LCIs for tomato cultivation are modelled similarly to kale production; thus, specific information on listed processes and data sources have been previously provided in Section 2.1.7.3.2.

The main characteristics and resource flows from tomato cultivation for farms in Georgia, USA and England, UK are provided in Table 28 and Table 31, respectively. Conventional tomato production varies drastically between the U.S. and UK. In Georgia, tomatoes are produced in open fields during two main growing seasons (summer and autumn). In the UK, temperatures are generally not warm enough for large-scale field production of tomatoes. Thus, tomatoes are typically grown under protective cover, usually using hydroponic production in heated glasshouses (Defra, 2019). The main growing season for UK conventional production is from January-November, with some plants also grown using indoor lighting from August-July, to allow for year-round production. Coconut coir and rockwool are used as the main soilless growing media, contained within plastic substrate bags into which the tomato crops are planted. Nutrient solution and water are passed to the crop using mainly drip irrigation (as is the case for this LCA), or nutrient film technique in some other cases.

Organic production in both countries is soil-based. For the U.S. farms considered, production takes place both outdoors and in polytunnels (plastic tunnels); however, for the UK farms, all organic production is performed in polytunnels. Irrigation is almost always applied in both cases, either through drip systems, sprinklers, or hand watering. Heating is only used by two farms, which include US-R-O-3 (using propane) and conventional UK tomato production (using natural gas). Other propane use is seen for US-PU-O-2, but this is from flame weeding. Other fuel uses are similar to those described for kale cultivation (see: Section 2.1.7.3.2).

The main electricity use across most farms is for irrigation purposes. However, additional electricity use for lighting is considered for UK conventional production. Additional considerations regarding energy use are required for UK conventional production, as two cases of conventional production are considered, which differ in the energy sources used. This is discussed in detail within the following section (2.1.7.4.3.1). For R-C-NG, electricity use is considered from the Great Britain (GB) national grid, as modelled within ecoinvent v.3.7.1. Natural gas use for heating glasshouses is considered for both R-C-NG and R-C-CHP, although in different amounts, as specified in Table 31. This is calculated based on the heating requirement of the glasshouse and the thermal efficiencies of the heating systems considered for each case; this assumes heating via a boiler system for R-C-NG and heating via combined heat and power for R-C-CHP. Detail on how the natural gas requirement was calculated for both cases is provided in Appendix A (see: Equation 30).

For R-C-CHP, the CHP system requires a higher natural gas input as both heat and electricity are produced from this system. The electricity requirement for R-C-CHP is the same as that considered for R-C-NG (11.2 kWh per gross m^2 cultivation area). Because the heating requirement is higher than the electricity requirement in the glasshouse, the natural gas used to satisfy this heating requirement results in the production of surplus electricity (300 kWh per gross m^2). This surplus electricity is assumed to be sold back to the national grid. Thus, the amount of natural gas used by the CHP system as designated in Table 31 (66.5 m^3 m^{-2}) is not fully allocated to the tomato crop lifecycle. The allocation of natural gas for heating and electricity within CHP differs among the allocation scenarios employed in this LCA, using energy allocation for Scenarios 1 and 2 and considering the avoided burdens of surplus electricity in Scenario 3. Further detail on how surplus electricity has been calculated and how burdens are allocated is provided in Section 2.1.7.4.3.2.

Fertility and pesticide inputs for tomato cultivation on U.S. and UK farms are provided in Table 29 and Table 32, respectively. Like for kale production, organic farms in the U.S. tend to use a wider range of fertiliser inputs, including manure, composts, seaweed / kelp extracts, fish emulsions, feathermeal and bloodmeal, and plant-based meals, like okraseed meal and alfalfa meal. On the other hand, most UK organic farms use primarily compost and manure as the main fertility inputs. Cover crops are also used among various organic and conventional farms in the U.S. and UK, although cover crops are never used as the only fertility input for tomato production, as was sometimes the case for kale. Both U.S. and UK conventional farms rely on mineral NPK fertilisers, although for the latter case, these will be applied as soluble fertilisers through drip irrigation systems. More detail regarding the specific nutrient and fertiliser inputs for UK conventional production is provided in Appendix A, Table 72.

Regarding pesticide products, organic pesticide use is common among U.S. organic farms, but not generally used on UK organic farms. As for kale cultivation, fumigation is practiced on U.S. conventional tomato farms. This involved fumigants (mainly chloropicrin) being applied to the soil using stake injectors. At the same time, beds are formed using tractors, drip tape is laid for irrigation, and then this is covered with white plastic mulch to ensure proper fumigation. This is left three weeks before transplanting tomato seedlings in line with U.S. regulations.

On the other hand, because UK conventional production occurs within a controlled environment (glasshouses), pest and disease can largely be regulated through strict hygiene and sanitation measures. When needed, biological or organic pesticide products will be applied, and biocontrol measures (predatory insects) are also used. Due to limited available information on the production of predatory insects, only energy and transport burdens are considered, based on data used within Williams, Audsley and Sandars (2006), as provided by the first author. Further information on the production burdens and specific active ingredients considered for UK conventional production is provided in Appendix A.2.

Finally, material and equipment use for U.S. and UK tomato cultivation is provided in Table 30 and Table 33, respectively. Generally, a higher amount of material use is considered for tomato cultivation relative to kale because of the material required for trellising tomatoes as vining crops. For production occurring in glasshouses and polytunnels, tomatoes are trellised to steel wires or hooks running the length of the structure. Plastic clips may also be used to attach the twine to the tomato stalks. For outdoor field-based production, there are no overhead structures to tie the twine to, and thus stakes or cages are required. In particular, U.S. organic farms may use steel cages (e.g., from hogwire fencing) or steel posts (e.g., T-posts). For U.S. conventional production, wooden pine stakes are used, which for the Georgia farms considered were chemically treated and mainly imported from Honduras. Tomatoes are then grown in these rows using staked trellis systems, typically using the common Florida weave system (UGA Extension, 2017; Kemble, 2019). For U.S. production, polypropylene twine is generally used, whilst for UK conventional production, twine made of natural fibres (jute) is used, so that this can be composted later on site. Organic farms across both countries may use either of these materials.

Other common material use is for irrigation, which is employed across all farms. U.S. conventional production utilises drip tape (LDPE film), whilst UK conventional production utilises a drip system, where drip stakes (also known as ‘pickaxes’) are placed into the substrate bags. Drip tubing is modelled for UK conventional production within Table 33, but mainlines used for drip irrigation are not included as they are modelled within the utilised AGRIBALYSE v.3.0 process. Organic farms also commonly employ drip irrigation systems, with a few also using sprinkler systems. Cultivation machinery is also included within Table 30 and Table 33. However, none is considered for UK conventional tomato production, as this is soilless production. On the organic farms, generally less machinery is used for tomato production than kale production, since often the majority of production occurs within polytunnels. However, small two-wheel tractors may be used to rotavate or till the soil. For U.S. conventional tomato production, a variety of tractors and equipment are used for cultivating the land, forming beds, laying plastic mulch and drip tape, and harvesting the crops. In all cases, harvesting occurs by hand, although tractors may be used to transport crops across fields.

Table 28 – U.S. tomato cultivation characteristics and major resource flows

Flows	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Gross area	m^2	372	261	146	456	268	1,085	326	323,749	1,048,406
Percentage as cropped area	%	67%	50%	71%	100%	64%	77%	50%	75%	75%
Average growing cycle [a]	weeks	25	20	26	32	21	24	32	21	15
Total harvest season [a]	weeks	17	13	22	22	8	34	26	14.5	17
Production in field	%	56%	77%	0	100%	0	0	21%	100%	100%
Production inside	%	44%	23%	100%	0	100%	100%	79%	0	0
Polytunnel type [b]	Type	US-P-4	US-P-5	US-P-3	n/a	US-P-4	US-P-1	US-P-1 (35%) US-P-2 (65%)	n/a	n/a
Planting density	Plants (cropped m^{-2})	0.71	2.01	6.41	1.91	3.51	2.87	2.54	0.95	1.51
Produced crop [c]	kg (gross m^{-2})	6.3	4.0	1.5	0.4	1.8	3.0	4.3	4.9	5.3
Harvested crop [c]	kg (gross m^{-2})	4.7	3.7	1.3	0.4	1.6	2.9	3.4	3.4	5.1
Harvest waste [c]	%	25%	8.7% [f]	10% [d]	7.1% [d]	12.5% [e]	0.5% [g]	20% [g]	30%	4%
Resources (gross m^{-2})										
Energy use, GA mix	kWh	0.389	0	0	0	0.822	0.049	1.43	0.046	0
Energy use, solar	kWh	0	0	0	0	0	0	1.33	0	0
Diesel use	L	0.006	0	0	0	0.007	0.003	0	0.026	0.034
Petrol use	L	0	0.052	0.038	0.006	0.008	0	0.619	0	0
Propane use	kg	0	0	0.034	0	0	0	0.681	0	0
Irrigation water use	L	169	261	124	0	222	112	697	356	120
Pesticide spray water use	L	0.11	0	0	0	0.11	0.04	2.90	1.51	1.05
Shed area, allocated [h]	m^2a	0	0	0	5.9E-04	2.1E-02	9.4E-04	0	8.6E-05	0

[a] The average growing cycle per planting is calculated with a weighted average of all plantings, based on area. The total harvest time may be longer than the average planting time if plantings are staggered.
[b] See Table 40 for LCIs for polytunnel infrastructures.
[c] Produced crop refers to the total crop that was grown in the field, based on farmer estimates or measurements. Harvest is the amount of crop that was harvested from the field. Harvest waste percent is then calculated as the total weight of crop wasted during harvest (i.e., sorted out in the field) divided by the total produced crop. All harvest waste is assumed to be returned to the soil unless otherwise denoted.
[d] This denotes harvest waste that is composted, or [e] fed to animals on the farm. [f] In this case half is turned into the soil and half is fed to animals, and [g] in this case half is composted and half is fed to animals.
[h] The shed area used to house cultivation equipment has been allocated to tomatoes based on time of use and / or weight or area allocation, depending on which is more appropriate. Note that the amount of shed space modelled here is only that specified by the farmer; when the farmer was unable to provide an estimate of this, shed space has been modelled using ecoinvent databases and is not reproduced here.

Table 29 – Fertility and pesticide inputs for U.S. tomato cultivation

Inputs per gross m² cultivated area	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Cover crop area, allocated to tomatoes	$m^2a\ m^{-2}$	0.231	0.026	0	0	0	0	0.039	0	0.313
Cover crop type (% by seed weight in mix)	type	Oat (40%), Pea (40%), Rye (20%)	Wheat (100%)	n/a	n/a	n/a	n/a	Rye (100%)	n/a	Wheat (100%)
Cover crop seed per area of cover crop, allocated to tomatoes	$kg\ m^{-2}$	0.006	0.052	0	0	0	0	0.001	0	0.007
Compost, purchased	$kg\ m^{-2}$	18.9	0.56	4.66	0	5.11	0.013	1.25	0	0
Compost, homemade	$kg\ m^{-2}$	0	0	0	0.08	0	0	0.28	0	0
Fresh manure	$kg\ m^{-2}$	0	0.278	0	0.535	0	0	0	0	0
Dried, pelleted manure	$kg\ m^{-2}$	0	0	0.028	0	0.032	0	0	0	0
Feather meal	$kg\ m^{-2}$	0.002	0	0	0	0.083	0.100	0	0	0
Kelp / seaweed meal & fish emulsion	$kg\ m^{-2}$	0.018	0	0.007	0	0	0.001	0.100	0	0
Other plant-based meals	$kg\ m^{-2}$	0	0.083	0	0	0.226	0.100	0	0	0
Peat moss	$kg\ m^{-2}$	0	0	0	0	1.41	0	0	0	0
Lime, as $CaCO_3$	$kg\ m^{-2}$	0.026	0	0	0	0	0	0	0.075	0.168
Other organic fertiliser mixes	$kg\ m^{-2}$	0.018	0.047	0	0	0.003	0	0.209	0	0
Mineral NPK fertilisers	$kg\ m^{-2}$	0	0	0	0	0	0	0	0.324	0.267
Total N, from cover crop residues	$g\ m^{-2}$	2.70	0.254	0	0	0	0	0.325	0	2.85
Total N, from fertilisers	$g\ m^{-2}$	191	16.5	48.6	11.7	75.1	16.1	28.0	21.3	23.2
Soluble N, from fertilisers	$g\ m^{-2}$	0.41	2.59	2.50	1.29	1.96	0.92	1.79	21.3	23.2
P_2O_5, from fertilisers	$g\ m^{-2}$	47.3	9.65	13.0	1.62	17.7	3.03	15.18	16.8	22.6
K_2O, from fertilisers	$g\ m^{-2}$	54.8	8.30	14.4	4.05	20.1	2.14	11.7	28.6	71.2
Biological pesticides, active ingredient (ai)	$g\ ai\ m^{-2}$	0.514	0	0	0	0.078	0.007	0.034	0	0
Other organic pesticides, active ingredient (ai)	$g\ ai\ m^{-2}$	0	0	3.32	0	0	0	50.7	0	0
Pesticides, active ingredients (ai)	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	10.5	15.2
…Insecticides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.101	0.185
…Herbicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	0.17	0.00
…Fungicides	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	1.67	3.12
…Fumigants	$g\ ai\ m^{-2}$	0	0	0	0	0	0	0	8.51	11.82
Compost transport [a]	km	282	219	241	0	340	663	161	0	0
Fresh manure transport [a]	km	0	0	0	2	0	0	0	0	0
Other input transport [a]	km	944	517	500	13	522	700	700	44	169

[a] This is the average one-way distance that compost and other inputs travel from the place of sale / collection to the farm. This value does not include other transport, such as from manufacturing centres to shops.

__Table 30__ – Materials and equipment for U.S. tomato cultivation, in g (gross m^{-2})

Item	Material	Process	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Shadecloth / landscape fabric	PP or HDPE	Extrusion and weaving	128 (4) [a]	0	0	0	2.83 (20)	12.3 (6)	985 (12)	0	0
Plastic mulch / tarps	LDPE	Extrusion, film	12.6 (3)	0	0	0	0	17.9 (1)	0	9.41 (1)	27.3 (1)
Drip tape	LDPE	Extrusion, film	23.9	2.41	0	0	0.788 (10)	2.03 (4)	7.87 (2)	2.15 (2.5)	9.75 (1)
Irrigation mainline	LLDPE	Extrusion, pipe	0.654 (4)	0	0	0	0.084 (20)	0.079 (15)	0	0	0.109 (10)
Irrigation mainline	PVC	Extrusion, pipe	0	0.26 (40)	0	0	0	0	0.176 (40)	0.181 (40)	0
Harvest crates	PP or HDPE	Injection moulded	0.426 (2)	0.06 (15)	1.15 (7)	0.062 (10)	0.091 (20)	0.328 (10) [b]	0.318 (10) [b]	0	0.013 (5)
Twine	PP or nylon	Extrusion and weaving	1.730 (3)	0.21 (5)	2.28 (7.5)	0.947 (1)	5.68 (3)	5.02 (1)	2.74 (2)	3.50 (1)	1.26 (1)
Trellis clips	PP	Injection moulded	2.23 (2.5)	1.31 (2)	0	0	0	0	9.09 (2)	0	0
Trellis hooks	Steel	Wire drawing	6.31 (5)	0.48 (20)	6.23 (20)	0	0	0	0	0	0
Wire / cages / posts	Steel	Wire drawing / section bar roll	33.7 (30)	77.7 (30)	0	49.7 (30)	33.9 (30)	26.3 (30)	156 (30)	0	0
Stakes	Wood	Sawnwood	0	0	0	0	0	0	0	129 (5)	106 (3)
Shipping containers for equipment store	Steel	Metal working, welding	14.5 (40)	0	0	0	0	0	0	0	0
Tractor, 2-wheel [c]	Mostly steel	Varies	0.546 (3,000 hr)	0.40 (3,000 hr)	0	0.058 (3,000 hr)	0	0	0	0	0
Tractor, small (<50 HP) [c]	Mostly steel	Varies	0	0	0	0	0.259 (9,000 hr)	0.107 (9,000 hr)	0	0	0
Tractor, medium (50-100 HP) [c]	Mostly steel	Varies	0	0	0	0	0	0	0	0.618 (10,000 hr)	0.098 (7,500 hr)
Tractor, large (>100 HP) [c]	Mostly steel	Varies	0	0	0	0	0	0	0	0.086 (10,000 hr)	1.62 (7,500 hr)
Implements [c]	Mostly steel	Varies	0.079 (800 hrs)	0	0	0	0	0.044 (800 hrs)	0	1.42 (varies)	1.06 (varies)
Mowers [c]	Mostly steel	Varies	0.132 (1,650 hrs)	0	0	0	0	0	0	0	0

[a] All material amounts are provided in g (gross m^{-2}), and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated.
[b] These materials have been reclaimed, and thus only transport and end of life waste burdens are attributed to these materials; no production burdens are considered.
[c] Tractors are designated based on horsepower (HP) rating. All tractors, implements, and other cultivation equipment are modelled using ecoinvent processes and thus specific material components are not provided in accordance with copyright principles. Amounts have been allocated to tomatoes based on the time of use for the crop, out of the total equipment lifetime, in hours. Lifetimes are estimated first from farmer data, then from secondary databases and literature sources.

Table 31 – UK tomato cultivation characteristics and major resource flows

Flows	Unit	U-O-1	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-CHP R-C-NG
Gross area	m^2	94.8	225	54.6	142	26.0	113	210	219	235,000
Percentage as cropped area	%	54%	80%	54%	87%	40%	71%	43%	37%	100%
Average growing cycle [a]	weeks	28	26	24	26	22	22	30	26	50
Total harvest season [a]	weeks	17	14	13	12	13	9	17	17	39
Production in field	%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Production inside	%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Polytunnel type [b]	Type	UK-P-2	UK-P-1	UK-P-2	UK-P-3	Reclaimed	UK-P-2	UK-P-1	UK-P-1	Glasshouse
Planting density	Plants (cropped m^{-2})	3.50	3.61	3.42	1.29	9.61	3.13	4.00	3.70	2.13
Produced crop [c]	kg (gross m^{-2})	1.39	3.69	4.95	1.94	1.44	2.84	6.03	3.89	34.5
Harvested crop [c]	kg (gross m^{-2})	1.29	3.38	4.21	1.73	1.44	2.67	6.03	3.34	34.3
Harvest waste [c]	%	7.57%[e]	8.43%[e]	15.0%[e]	10.9%[d]	0	6.25%[d]	0	14.1%[d]	0.547%[d]
Resources (gross m^{-2})										
Electricity use, GB mix	kWh	0	0	0	0	0	0	0.021	0	11.2 for NG
Surplus electricity from CHP	kWh	0	0	0	0	0	0	0	0	300 [f]
Natural gas use	m^3	0	0	0	0	0	0	0	0	33.5 for NG 66.5 for CHP [f]
Diesel use	L	0	0	0	0.008	0	0	0.005	0.029	0.006
Petrol use	L	0	0	0.235	0	0	0	0.007	0.009	0
Irrigation water use	L	88.5	141	1,757	56.5	0.00	71.1	69.4	99.0	1,143
Irrigation water use, rainwater [g]	L	0	0	0	0	38.4	0	69.4	0	0
Pesticide / spray water use	L	0.110	0	0	0.00	0.113	0.042	2.90	1.51	0.11
Shed area, allocated [h]	m^2a	0	0	0.0002	0	0	0	0	0.0014	0

[a] The average growing cycle per planting is calculated with a weighted average of all plantings, based on area. The total harvest time may be longer than the average planting time if plantings are staggered.

[b] See Table 40 for LCIs for polytunnel infrastructures, listed for labelled infrastructures. Large-scale glasshouses are modelled using AGRIBALYSE and not reproduced to maintain copyright principles. Reclaimed tunnels are made out of waste materials, for which production burdens are not attributed; thus, inventories have not been listed for these tunnels.

[c] Produced crop refers to the total crop that was grown in the field, based on farmer estimates or measurements. Harvest is the amount of crop that was harvested from the field. Harvest waste percent is then calculated as the total weight of crop wasted during harvest (i.e., sorted out in the field) divided by the total produced crop. All harvest waste is assumed to be returned to the soil unless otherwise denoted.

[d] This denotes harvest waste that is composted; [e] in this case, approximately half of the waste is turned into the soil and half is composted.

[f] Surplus electricity is only considered for R-C-CHP. Allocation of surplus electricity and natural gas use for CHP differs based on the employed allocation Scenario, with details provided in Section 2.1.7.4.3.

[g] This refers to harvested rainwater used for irrigation. This is included for informational purposes only; harvested rainwater is not counted as water consumption within the utilised impact assessment method.

[h] The shed area used to house cultivation equipment has been allocated to tomatoes based on time of use and / or weight or area allocation, depending on which is more appropriate. Note that the amount of shed space modelled here is only that specified by the farmer; when the farmer was unable to provide an estimate of this, shed space has been modelled using ecoinvent databases and is not reproduced here.

Table 32 – *Fertility and pesticide inputs for UK tomato cultivation*

Inputs, per gross m^2 cultivated area	**Unit**	**U-O-1**	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-CHP R-C-NG**
Cover crop area, allocated to tomatoes	m^{2}a m^{-2}	0.017	0	0	0	0	0	0.500	0	0
Cover crop type (% by seed weight in mix)	type	Rye (30%); clover (30%), other (40%)	n/a	n/a	n/a	n/a	n/a	Rye (76%), vetch (24%)	n/a	n/a
Cover crop seed per cover cropped area, allocated to tomatoes	kg m^{-2}	7.8E-05	0	0	0	0	0	0.0014	0	0
Compost, purchased	kg m^{-2}	3.70	20.0	5.49	0	0	0	3.47	0	0
Compost, homemade	kg m^{-2}	0.566	0.667	5.49	2.12	11.4	2.03	0	0	0
Fresh manure	kg m^{-2}	3.14	0	0	0	0	0	3.30	2.34	0
Coco coir, growing media	kg m^{-2}	0	0	0	0	0	0	0	0	0.582
Rockwool, growing media	kg m^{-2}	0	0	0	0	0	0	0	0	0.041
Other organic fertiliser mixes	kg m^{-2}	0	0.035	0.238	0	0.384	0	0	0	0
Mineral NPK fertilisers	kg m^{-2}	0	0	0	0	0	0	0	0	1.09
Mineral micronutrient fertilisers	kg m^{-2}	0	0	0	0	0	0	0	0	0.614
Total N, from cover crop residues	g m^{-2}	0.810	0	0	0	0	0	16.1	0	0
Total N, in all fertilisers	g m^{-2}	47.5	169	97.2	17.16	400	14.7	48.1	9.57	125
Soluble N, in all fertilisers	g m^{-2}	4.12	7.69	9.98	0.77	18.4	0.73	3.27	1.92	125
P_2O_5, in all fertilisers	g m^{-2}	18.8	68.9	41.0	6.99	47.8	6.02	19.7	3.51	56.6
K_2O, in all fertilisers	g m^{-2}	46.4	138	86.8	0.01	78.6	11.6	37.8	13.6	237
Biological pesticides, active ingredient (ai)	g ai m^{-2}	0	0	0	0	0	0	0	0	0.740
…Biological insecticides	g ai m^{-2}	0	0	0	0	0	0	0	0	0.590
…Biological fungicides	g ai m^{-2}	0	0	0	0	0	0	0	0	0.150
Biocontrol (predatory insects)	number m^{-2}	0	0	0	0	0	0	0	0	88.7
Bumblebee colonies	number m^{-2}	0	0	0	0	0	0	0	0	0.006
Energy use, biocontrol and pollinator production [a]	kWh m^{-2}	0	0	0	0	0	0	0	0	2.78
Compost transport [b]	km	22	22	46	0	0	0	15	0	0
Fresh manure transport [b]	km	11	0	0	0	0	0	11	4	0
Other input transport [b]	km	11	117	75	0	2	0	13	4	994

[a] The production of predatory insects and pollinators is approximated based on the energy used for production, as used in Williams, Audsley and Sandars (2006), with figures provided by the authors.
[b] This is the average one-way distance that compost and other inputs travel from the place of sale / collection to the farm. This value does not include other transport, such as from manufacturing centres to shops.

***Table 33** – Materials and equipment for UK tomato cultivation, in g (gross m^{-2})*

Item	**Material**	**Process**	**U-O-1**	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-CHP R-C-NG**
Plastic for substrates	LDPE	Extrusion, film	0 [a]	0	0	0	0	0	0	0	12.3 (1 use)
Plastic mulch / sheets	LDPE or PP	Extrusion, film	0.430 (8)	0	0	0	10.8 (18)	10.0 (10)	15.2 (1)	0	34.2 (2)
Plastic mulch	PLA	Extrusion, film	0	0	10.7 (1 use)	0	0	0	0	0	0
Drip tape	LDPE	Extrusion, film	0	0	3.77 (3.5)	0	0	1.93 (3)	0	2.92 (3)	0
Drip tubes / pipes	PE	Extrusion, pipe	0	0	0	0	0.013 (20)	0.092 (30)	1.07 (30)	0	8.36 (5)
Drip stakes	PP or HDPE	Injection moulded	0	0	0	0	0	0	0.125 (30)	0	0.354 (5)
Irrigation mainline	PVC	Extrusion, pipe	0	4.74 (40)	0.269 (7.5)	0	0	0	0.100 (40)	1.16 (40)	0
Soaker hose	Rubber	Extrusion, pipe	8.56 (7)	0	0	6.36 (10)	0	0	0	0	0
Sprinklers	PP or HDPE	Injection moulded	0	0.500 (18)	0	0	0	0	0	0.160 (30)	0
Harvest crates	PP or HDPE	Injection moulded	0.185 (25) [b]	0.383 (25) [b]	0.470 (3) [b]	0.324 (2) [b]	0.598 (25) [b]	0.215 (25) [b]	0.280 (15) [b]	0.093 (10) [b]	16.53 (15)
Twine	PP	Extrusion and weaving	3.67 (4 uses)	7.33 (2.5 uses)	0	3.59 (2.5 uses)	3.49 (3.5 uses)	5.04 (3.5 uses)	5.44 (2 uses)	4.63 (2 uses)	0
Twine	Jute fibre	Weaving	0	0	4.16 (2)	0	0	0	0	0	881 (1)
Wire & trellis hooks	Steel	Wire drawing	0	1.71 (30)	1.14 (30)	1.60 (30)	1.20 (30)	0.520 (30)	1.10 (30)	0.240 (30)	50.5 (7)
Insect sticky traps	HDPE	Extrusion, film	0	0	0	0	0	0	0	0	0.032 (1)
Pond liner	Butyl	Thermoforming	0	0	0	0	0	0	0.502 (15)	0	0.108 (15)
Rainwater tanks	HDPE	Injection moulded	0	0	1.54 (30)	0	32.8 (20)	0	0	0	0
Rainwater tanks	Steel	Section bar rolling	0	0	0	0	86.1 (20)	0	2.12 (15)	0	0
Shipping containers for equipment store	Steel	Metal working, welding	0	0	0	7.77 (40)	0	1.96 (40)	0	0	0
Tractor, 2-wheel [c]	Mostly steel	Varies	0	0	0	0.659 (38)	0	0	0.571 (40)	0.030 (3,000 hr)	0
Tractor, small (<50 HP) [c]	Mostly steel	Varies	0	0	0	0	0	0	1.37 (9,000 hr)	1.01 (9,000 hr)	0
Implements [c]	Mostly steel	Varies	0	0	0	0.517 (800 hr)	0	0	0	0.413 (30)	0
Mowers [c]	Mostly steel	Varies	0	0	0	0	0	0	0.037 (750 hr)	0	0

[a] All material amounts are provided in g (gross m^{-2}), and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated.
[b] These materials have been reclaimed, and thus only transport and end of life waste burdens are attributed to these materials; no production burdens are considered.
[c] All tractors, implements, and other cultivation equipment are modelled using ecoinvent processes and thus specific material components are not provided in accordance with copyright principles. Amounts have been allocated to tomatoes based on the time of use for the crop, out of the total equipment lifetime, in hours. Lifetimes are estimated first from farmer data, then from secondary databases and literature sources.

2.1.7.4.3 Additional considerations for UK conventional tomato production

There are several additional considerations required for UK conventional tomato production. Because hydroponic production differs so greatly from soil-based systems, different materials and inputs are required for this production type. Additionally, because of the protective nature of the controlled environment agriculture industry, obtaining primary data from UK conventional growers was difficult. In most cases, only general information or industry averages were given; in some cases, such as for nutrients and fertilisers used, the data could not be shared. Thus, the LCI for UK conventional tomato production relies on significantly more secondary and literature data than any other crop LCI. Appendix A explains sources of data, uncertainties, and assumptions made for this LCI in more detail.

2.1.7.4.3.1 Energy systems

Additionally, for UK conventional tomato production, two systems for heating and electricity are considered based on what is commonly used by UK tomato growers. This was done to provide additional context to the impacts of UK conventional tomato production, since only one farm case study was provided in this category. Additionally, prior research has identified that the lifecycle impact assessment for UK conventional tomato production is very sensitive to the heating system used (Williams, Audsley and Sandars, 2006), so alternative scenarios were investigated.

In this study two cases are considered: 1) tomato production using the conventional energy supply, which includes electricity from the national grid and natural gas heating through a boiler system (R-C-NG), and 2) tomato production using a combined heating and power system (CHP) at the farm site (R-C-CHP). These two energy systems are only considered for the heating and electricity supply needed for glasshouse operation, and thus are only distinct within the germination and cultivation phases of production. The processing and transport stages both assume electricity usage from the national grid.

Combined heat and power (CHP), also called co-production, is becoming a more common method of producing heat and electricity for horticultural production in heated glasshouses in the UK. A CHP system consists of an electrical generator combined with equipment that allows for the heat produced by the generator to be recovered (BEIS, 2021b). The benefit of the CHP system is that it generally produces heat and electricity at much higher efficiencies than when these services are provided separately (U.S. Department of Energy, 2022). The modelling of CHP systems in this LCA is based on information provided by the grower and the nursery, which both specified that CHP is used as the energy source. Thus, R-C-CHP represents the actual site-specific growing conditions.

The models of each energy system include fuel input, infrastructure, and relevant emissions (ecoinvent Centre, 2021). For the first case of conventional energy, ecoinvent v.3.7.1 processes are used to model electricity supply from the Great Britain national grid and centralised heat supply from a boiler, using natural gas (ecoinvent Centre, 2021). For the second case, the ecoinvent v.3.7.1 process for combined heat and power production through a gas engine is used as a base process, but is updated to reflect specific natural gas usage and emissions for the type of CHP system used on the farm, as described in the following paragraphs.

For the CHP system, surplus electricity is produced as an output since the heat used is in excess of the electricity requirement. The surplus is calculated as the difference between the electricity produced by the CHP unit whilst satisfying the heat demand and the electricity requirement for tomato production (11.2 kWh m^{-2}), yielding an approximate surplus of 300 kWh m^{-2}, as designated in Table 31. Further detail on the calculations of surplus electricity, based on heating requirements and amounts of natural gas used, is provided in Appendix A (see: Equation 32). This surplus electricity is considered to be sold back to the grid; thus, allocation is required so that only the amount of natural gas needed to satisfy the heat and electricity requirements for tomato glasshouse production is considered.

2.1.7.4.3.2 Allocations of surplus electricity from CHP

Two types of allocation are applied to surplus electricity production from CHP: allocation based on energy content, used within cut-off allocation scenarios (1 and 2), and avoided burdens of the electricity supply from the grid, used within the system expansion scenario (3).

The energy allocation scenario requires further calculations, which are based upon those used by Blonk et al. (2010). In this method, the input of natural gas, as specified in Table 31 for R-C-CHP, is allocated to heat and electricity based on thermal and electrical efficiencies of the CHP system, respectively, as well as the heating efficiency of the glasshouse. To determine the input of natural gas that is allocated to the production of electricity, Equation 1 should be used.

$$E_a = \frac{e_{eff}}{(e_{eff} + (t_{eff} * h_{eff}))}$$

Equation 1 – Allocation of CHP burdens to electricity production

Where,

- E_a = allocation factor for electrical energy from CHP, as [%]
- e_{eff}= the electrical efficiency of the CHP unit, as [%]
- t_{eff} = the thermal efficiency of the CHP unit, as [%]
- h_{eff} = the heating efficiency of the glasshouse, as [%]

Following this, the allocation to heat produced from the CHP unit would be 1-E_a (with E_a in decimal form).

The efficiencies of the CHP system, as provided by the supplier used by the case study farm, are 42.5% for electrical efficiency and 46.5% for thermal efficiency. The heating effiency of the glasshouse is assumed as 96%, based on that achieved by heated glasshouses in the Netherlands (Blonk *et al.*, 2010). Further detail on these efficiencies and associated uncertainties are provided in Appendix A, Table 70. Using these efficiencies, the allocation to electrical energy for this study can be calculated as 48.77% and that to heat as 51.23%. The electrical allocation is slightly higher than the value of 45.5% used by Blonk et al. (2010), due to the higher electrical efficiency of the CHP system used within this study, although similar. The allocation factors for heat and electricity can be directly applied to the amounts of heat and electricity used by the glasshouses, by multiplying the amounts used by their relevant allocation factors. An alternative way of approaching this would be to apply the electrical allocation factor to the amount of surplus electricity produced and subtract this from the total amount of heat used. Both cases yield the same results.

For the system expansion scenario, avoided burdens are attributed by using the electricity supply from the national (Great British) grid as the alternative scenario. Thus, the burdens of producing the amount of surplus electricity from the Great British grid are subtracted from the total burdens of CHP production (e.g., total consumption used to satisfy both heating and electricity requirements). The national grid electricity production mix is based on the year 2017 and is modelled using ecoinvent v.3.7.1 processes (ecoinvent Centre, 2021).

2.1.7.4.3.3 Emissions from CHP

The gaseous emissions associated with burning natural gas in the CHP system are based on emission factors provided by the European Environment Agency (EEA) guidance (EMEP & EEA, 2019c) and the UK Government (BEIS, 2021a), specifically defined for reciprocating gas engines like the one used on the farm of interest. All emission values used are based on gross calorific value of natural gas, scaled using a gross:net calorific value ratio of 1.1 (EMEP & EEA, 2019c; BEIS, 2021b).

2.1.7.4.4 Processing and storage

The LCIs from processing and storage of tomatoes on the farm (or at relevant farm packhouses) are considered within this section, listed per kg sellable crop. Table 34 and Table 36 provide an overview of major resource flows during tomato processing and storage for U.S. and UK farms, respectively, as well as listing processing waste amounts (out of total harvest) and infrastructure use, such as packhouses and cold storage facilities. Tomato waste during processing is generally a result of split fruit or other cosmetic issues. Details on how waste is managed on each farm is provided in the relevant footnotes of each table and is similar to that for kale. Only for the UK conventional farm (R-C-NG / R-C-CHP) is processing waste landfilled. Unlike for kale, tomatoes are generally not washed in either the U.S. or UK; thus, few farms will have any water use flows. Table 35 and Table 37 then provide an overview of the materials used to package and store the crop before final distribution per kg sellable crop, for U.S. and UK farms respectively. As for the kale LCIs, all material amounts and lifetimes are based on primary data provided by farmers or their relevant suppliers.

The general processes for packaging and storing tomatoes differ more among U.S. and UK conventional farms than organic farms. In the U.S., tomatoes are generally harvested directly into the cardboard boxes in which they are sold. For R-C-1, these boxes are often directly loaded onto pallets and then into lorries for transport the same day; thus, no storage of tomatoes occurs on the farm. For R-C-2, tomatoes are stored temporarily in cool stores, usually for only 24 hours before transport. Like for U.S. conventional kale, tomatoes are sold loose in cardboard boxes to wholesalers or directly to retailers; most supermarket chains considered again sell tomatoes loose rather than packaged, and thus no additional packaging is considered.

For UK conventional production, tomatoes are harvested into plastic storage crates and stored at a cool storage facility temporarily on the farm site. They are then sent to another packhouse for packaging, which serves several growing sites within the same business. The UK conventional tomato grower was also unable to provide primary data on energy and fuel use for tomato processing and storage within the farm and off-site packhouse; thus, electricity use for packhouse operations and cool storage, as well as for propane use in packhouse operations, has been estimated based on figures from other UK vegetable farmers in this study and from the PEFCR guidance for the cool storage of crops (European Commission, 2018). Further details of the estimations and assumptions made for this UK conventional tomato LCI are provided in Appendix A (see: Table 74).

Like for UK conventional kale, these UK conventional tomatoes are usually packaged into retailer-specific packaging, which also differs based on the tomato variety. These include simple plastic packs and plastic or paper punnets, with some varieties also sold loose. Packaging amounts were measured by weighing the packaging for relevant tomato types from the different supermarket retailers that the farm distributed to. The amounts of each packaging used was then estimated based on the weights normally packaged into each packaging type and the amounts of each tomato type produced, as provided by the grower (based on industry average data).

For organic farms in the U.S. and UK, tomatoes are generally not stored in cool storage facilities. In the U.S., due to the hot temperatures, fans or A/C units may be used to provide cooling for the crop, although this is not common in the UK. Two organic U.S. farms (R-O-2 and R-O-3) use air conditioning within insulated rooms to store their tomato crops, at temperatures of 13-15.5°C (55-60°F), with the prior farm using a CoolBot cooling systems as described in Section 2.1.7.2.8. Additionally, some farms in the U.S. (PU-O-1 and PU-O-2) use chest freezer or refrigerators to preserve their tomatoes for longer. In the UK, only PU-O-4 uses cool storage facilities for tomatoes. Additional information about modelling of cold storage infrastructure is provided in Section 2.1.7.5.2. For all other UK organic farms, energy use is considered as zero. It should be noted that most UK organic farm ‘packhouses’ are actually small sheds or barns that do not have any electricity.

Tomato packaging for U.S. and UK organic farms generally includes cardboard boxes or bags for farms selling to restaurants or in CSAs, or paper bags and paper punnets for those selling through markets or farm stands. In many cases, punnets and cardboard boxes are reused for the same crop over the season, especially as many consumers may bring their own bags for packaging (e.g., at farmer’s markets). Plastic bags may also be used in some cases, especially when selling multiple crops together (e.g., in CSAs or veg boxes); in this case, packaging has been allocated to tomatoes based on the weight of the crop out of the total weight of all crops in the bag or box.

***Table 34** – U.S. tomato processing and storage: major resource flows*

Flows	**Unit**	**U-O-1**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**R-O-1**	**R-O-2**	**R-O-3**	**R-C-1**	**R-C-2**
Sellable crop, total	kg	1,484	953	176	178	359	2,957	889	1,102,229	4,696,484
Sellable crop, per gross m^2	kg m^{-2}	3.99	3.65	1.21	0.39	1.34	2.73	2.73	3.40	4.48
Processing waste, % of harvest [a]	%	15% [b]	0%	10% [b]	0%	17% [c]	7.1% [d]	20%[b,c]	0%	12.5%[e]
					Resources (kg sellable crop)$^{-1}$					
Energy use, GA mix	kWh	0	0.041	0.043	0	0.587	0.091	0.057	0	0.018
Energy use, solar	kWh	0	0	0	0	0	0	0.057	0	0
Water use	L	0	0	0	51.0	0	0	0	0	0
Packhouse area, allocated [f]	m^2a	0.005	0.005	0.033	0	0.001	0.004	0	0	0
Ambient storage area, allocated [f]	m^2a	0.001	0	0	0.070	0.001	0	0	0	0
Cool storage area, allocated [f]	m^2a	0	0	0	0	0	0.001 [g]	0.002 [g]	0	1.46E-05
Fridge / freezer volume, allocated [h]	m^3a	0	5.9E-05	3.3E-04	0	0	0	0	0	0

[a] Processing waste is listed as the percent of crop wasted during processing and storage, out of the total harvested crop.
[b] This indicates processing waste that is composted; [c] fed to animals on the farm; [d] donated to local churches or charities; [e] or returned back to the field soil.
[f] Packhouse and storage areas may be steel or wooden shed structures. Space is allocated to tomatoes based either on the space used for the crop and the time used out of the year, or using weight allocation over the year (out of all crops produced on the farm), depending on which is more appropriate.
[g] This indicates cool storage areas that utilise air conditioning for cooling, rather than conventional refrigeration; this either uses traditional A/C units or A/C units in conjunction with the CoolBot cooling system (https://www.storeitcold.com/).
[h] These refer to standing refrigerators and chest freezers.

Table 35 – Materials for packaging and storage of U.S. tomatoes, in g (kg sellable crop)$^{-1}$

Item	Material	Process	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Cardboard boxes	Container-board	Box board production	0 [a]	0	0	0	22.5 [b] (1.5 uses)	9.08 (1 use)	18.8 (1 use)	42.2 (1 use)	52.0 (1 use)
Paper punnets	Boxboard / coreboard	Box board production	0	0	0	0	0	0	0	0	3.69
Paper punnets	Recycled coreboard	Box board production	6.56 (reused over season)[c]	0	17.9 (2 uses)	0	0	0	0	0	0
Paper bags	Recycled paper	Paper recycling, kraft paper production	0	0	0	0	16.2 (1 use)	3.69 (1 use)	7.30 (1 use)	0	0
Plastic bags	LDPE	Extrusion, film	0	0	0.33 (1 use)	0	0	0	0	0	0
Plastic bags	HDPE	Extrusion, film	0	0	3.02 (1 use)	0	0	0	0	0	0
Plastic bags	PLA	Extrusion, film	0	0	3.44 (1 use)	0	0	0	0	0	0
Storage crates	PP / LDPE / HDPE	Injection moulded	0.428 [b] (10)	0.140 (15)	1.51 (7)	0.316 (10)	0.909 (10)	0.192 (23)	0	0	0
Pallets	Wood; steel	Sawnwood; hot rolled	0	0	0	0	0	0	0	0.675 (30 uses)	0.606 (30 uses)
Shipping containers for crop storage	Steel	Metal working, welding	0	0	0	41.1 (40)	0	0	0	0	0
Average transport of packaging materials (km) [d]	n/a	n/a	16	3,753	310	44	990	272	79	134	505

[a] All material amounts are provided in g (kg sellable crop)$^{-1}$, and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated. If lifetime is designated in 'uses', it is assumed that one use is either 1 kg of crop packaged (in the case of packaging) or 1 kg crop transported, in the case of transport materials like pallets or crates.

[b] These materials have been reclaimed from other businesses, and thus only transport and end of life waste burdens are attributed to these materials, when applicable; no production burdens are considered.

[c] These punnets are reused just for tomatoes over the entire harvest season, so this considers the total amount used over the season, per kg sellable tomato.

[d] This is the average one-way transport distance of packaging materials, from the place of purchase to the farm or packhouse site, in km. This distance does not include other transport, such as from manufacturing centres to shops, which is accounted for within modelled ecoinvent processes.

***Table 36** – UK tomato processing and storage: major resource flows*

Flows	Unit	U-O-1	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-CHP R-C-NG
Sellable crop, total	kg	116	751	230	242	37.5	300	1217	734	7,859,743
Sellable crop, per gross m^2	kg m^{-2}	1.22	3.34	4.21	1.71	1.44	2.67	5.80	3.34	33.4
Processing waste, % of harvest [a]	%	5.0% [b]	1.2% [b,c]	0	1% [b]	0	0	4.0% [b,c]	0	2.5% [d]
Resources (kg sellable crop)$^{-1}$										
Energy use, GB mix	kWh	0	0	0	0	0	0.067	0	0	0.220
Propane use	kg	0	0	0	0	0	0	0	0	0.0052
Water use	L	0	0	0	0	0	0	0	0	0
Packhouse area, allocated [e]	m^2a	0.0012	0.0011	0	0	0.0297	0.0030	0.0308	0.0116	0.0003
Insulated storage area, allocated [e]	m^2a	0	0	0	0.0001[f]	0	0	0	0	0
Refrigerated cold storage area, allocated [e]	m^2a	0	0	0	0	0	3.33E-04[f]	0	0	3.02E-06

[a] Processing waste is listed as the percent of crop wasted during processing and storage, out of the total harvested crop.
[b] This indicates processing waste that is composted; [c] given to employees or eaten on the farm; [d] or landfilled.
[e] Packhouse and storage areas may be steel or wooden shed structures. Space is allocated to tomatoes based either on the space used for the crop and the time used out of the year, or using weight allocation over the year (out of all crops produced on the farm), depending on which is more appropriate.
[f] These both refer to shipping containers that are used for crop storage; for PU-O-2, this is just an insulated shipping container, but for PU-O-4, the shipping container includes a refrigeration system. Material use for the shipping container structures are also included in Table 37.

Table 37 *– Materials for packaging and storage of UK tomatoes, in g (kg sellable crop)*$^{-1}$

Item	Material	Process	U-O-1	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-CHP R-C-NG
Cardboard boxes	Container-board	Box board production	0 [a]	0	5.31 (6 uses)	10.2 [b] (6 uses)	0	0	0	0	0
Paper punnets	Boxboard / pulp	Box board production	0	0	0	0	0	0	0	31.3 (reused over season)[c]	32.2 (1 use)
Paper bags	Paper	Kraft paper production	0	15.9 (1 use)	0	0	0	12.5 (1 use)	0	0	0
Paper bags	Recycled paper	Paper recycling, paper production	0	0	0	0	14.6 (1 use)	0	0	0	0
Plastic punnets	PET	Injection moulded	0	0	0	0	0	0	0	0	5.79 (1 use)
Plastic film	PP	Extrusion, film	0	0	0	0	0	0	0	0	6.98 (1 use)
Plastic bags	LDPE	Extrusion, film	0.26 (1 use)	0	0	0	0	0	0	0	0
Biodegradable plastic bags	Biodegradable LDPE / PLA	Extrusion, film	0	0.071 (2)	0	0	0	2.73 (2.5 uses)	0	0	0
Reusable tote bags	Jute	Textile production	0	0	0	0	1.65 (23)	0	0	0	0
Storage crates	PP	Injection moulded	0.151 (25) [b]	0.115 (25) [b]	0	0	0	0.081 (25) [b]	0.048 (15) [b]	0	1.822 (75 uses)
Storage crates	HDPE	Injection moulded	0	0.006 (10) [b]	0.111 (3) [b]	0.189 (2) [b]	0	0	0	0.028 (10) [b]	0.494 (15)
Pallets	Wood; steel	Sawnwood; hot rolled	0	0	0	0	0	0	0	0	1.81 (30 uses)
Shipping containers for crop storage	Steel	Metal working, welding	0	0	0	0.461 (40)	0	1.15 (40)	0	0	0
Average transport of packaging material (km)[c]	n/a	n/a	8	50	26	0	258	185	0	217	136

[a] All material amounts are provided in g (kg sellable crop)$^{-1}$, and are already allocated based on time of use and material lifetime. Applied material lifetimes are provided in parentheses (), in years unless otherwise stated. If lifetime is designated in 'uses', it is assumed that one use is either 1 kg of crop packaged (in the case of packaging) or 1 kg crop transported, in the case of transport materials like pallets or crates.

[b] These materials have been reclaimed, and thus only transport and end of life waste burdens are attributed to these materials, when applicable; no production burdens are considered.

[c] These punnets are reused just for tomatoes over the entire harvest season, so this considers the total amount used over the season, per kg sellable tomato.

[d] This is the average one-way transport distance of packaging materials, from the place of purchase to the farm or packhouse site, in km. This distance does not include other transport, such as from manufacturing centres to shops, which is accounted for within modelled ecoinvent processes.

2.1.7.4.5 Distribution

The distribution phase for tomatoes is modelled similarly to that for kale (see: Section 2.1.7.3.4). Table 38 and Table 39 provide an overview of the tomato transport distances and resource use for crop storage in regional distribution centres (RDCs), per kg sellable crop, for U.S. and UK farms, respectively.

As for kale, longest average transport distances for tomatoes are seen on conventional farms in both countries, with the U.S. conventional farms mainly selling tomatoes regionally (in the southeast U.S.) and the UK conventional farm selling tomatoes in retail chains across Great Britain. Among the organic farms, the longest transport distances are seen for tomatoes sold from rural farms or farms selling through home delivery routes (UK-PU-O-1, UK-PU-O-2, UK-PU-O-4).

Modes of transport are similar to that seen for kale, with organic farms transporting crops mainly in passenger cars, whilst conventional farms transport crops mainly through large lorries (16-32 metric tonnes). For UK conventional production, this transport occurs through refrigerated lorries, which are accounted for within ecoinvent processes. Some organic farms in the UK also transport a portion of their crop using electric bikes, for which electricity usage has been estimated. More detail on the transport processes used and how transport is allocated for different modes of transport is provided in Section 2.1.7.2.8.

Storage burdens for crops travelling through regional distribution centres (RDCs) differs for U.S. and UK tomatoes; for the prior, ambient storage is assumed because this is how the farms tend to store their crop, and thus additional electricity use from cold storage is not considered. However, cold storage for tomatoes is considered for UK crops, as per the farmer.

__Table 38__ – U.S. tomato distribution: major resource flows

Flows	Unit	U-O-1	PU-O-1	PU-O-2	PU-O-3	R-O-1	R-O-2	R-O-3	R-C-1	R-C-2
Transported / sellable crop, total	kg	1,484	953	176	178	359	2,957	889	1,102,229	4,696,484
Average one-way trip distance [a]	km	20	0	11	0	91	69	40	469	617
Flows (kg sellable crop)$^{-1}$										
Transport via large passenger car, allocated [b]	km	0.232	0	2.32	0	3.65	1.20	1.54	0	0
Transport via lorry [c]	kg*km	0	0	0	0	0	0	0	469	617
…transport from farm to RDC	kg*km	0	0	0	0	0	0	0	340	360
…transport from RDC to shops	kg*km	0	0	0	0	0	0	0	129	257
RDC storage area (ambient storage) [d]	m^3a	0	0	0	0	0	0	0	4.84E-04	5.98E-04
Electricity use, RDC [d]	kWh	0	0	0	0	0	0	0	1.45E-04	1.79E-04
Heat use, RDC [d]	MJ	0	0	0	0	0	0	0	1.74E-03	2.15E-03
Water use, RDC [d]	L	0	0	0	0	0	0	0	5.90E-05	7.30E-05

[a] This is the average trip distance travelled by the tomato crop to all points of sale. It is calculated as a weighted average of all trip distances travelled. For organic farms, this is weighted based on the total number of trips; for conventional farms, this is weighted based on the proportion of crop sold at each sale point. It also considers crop that is sold on the farm as a '0 km' trip distance.
[b] Transport via passenger cars is modelled in ecoinvent in terms of km. Thus, the total round-trip distance travelled by all kale crop (summing each journey) is calculated and then allocated to kale based on its proportion out of all crops transported. This is the figure represented in this row.
[c] Transport via lorry is modelled in ecoinvent as kg*km. Thus, kg*km per kg sellable crop is the same as the average trip distance provided in the second row.
[d] This is calculated based on the volume of the packaged crop (e.g., box volume), * 4 for ambient crops, and assuming a storage of 4 weeks as per PEFCR guidance (European Commission, 2018). This value considers both the infrastructure for the RDC centre and ambient storage space, but not cold storage infrastructure. Electricity, heat, and water use is also based on PEFCR guidance.

Table 39 – *UK tomato distribution: major resource flows*

Flows	**Unit**	**U-O-1**	**U-O-2**	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-CHP R-C-NG**
Transported / sellable crop, total	kg	116	751	230	242	37.5	300	1217	734	7,859,743
Average one-way trip distance [a]	km	11	19	28	123	6	140	74	15	293
Flows (kg sellable crop)$^{-1}$										
Transport via large passenger car, allocated [b]	km	0.059	0.104	0.155	0.250	0.589	0	0	0	0
Transport via electric bike, allocated	km	0.527	0	0	0.005	0	0	0	0	0
Transport via diesel transit van	kg*km	0	0	19	94	0	140	74	0	0
Transport via lorry [c]	kg*km	0	0	0	0	0	0	0	0	293 [d]
…transport from farm to RDC	kg*km	0	0	0	0	0	0	0	0	196
…transport from RDC to shops	kg*km	0	0	0	0	0	0	0	0	97
RDC storage area (cold storage) [e]	m^3a	0	0	0	0	0	0	0	0	2.56E-04
Electricity use, RDC [e]	kWh	0	0	0	0	0	0	0	0	1.03E-02
Heat use, RDC [e]	MJ	0	0	0	0	0	0	0	0	9.21E-04
Water use, RDC [e]	L	0	0	0	0	0	0	0	0	3.12E-05

[a] This is the average trip distance travelled by the tomato crop to all points of sale. It is calculated as a weighted average of all trip distances travelled. For organic farms, this is weighted based on the total number of trips; for conventional farms, this is weighted based on the proportion of crop sold at each sale point. It also considers crop that is sold on the farm as a '0 km' trip distance.
[b] Transport via passenger cars is modelled in ecoinvent in terms of km. Thus, the total round-trip distance travelled by all kale crop (summing each journey) is calculated and then allocated to kale based on its proportion out of all crops transported. This is the figure represented in this row.
[c] Transport via lorry is modelled in ecoinvent as kg*km. Thus, kg*km per kg sellable crop is the same as the average trip distance provided in the second row.
[d] This transport occurs through refrigerated lorries.
[e] This is calculated based on the volume of the packaged crop (e.g., box volume), * 3 for chilled crops, and assuming a storage of 1 week as per PEFCR guidance (European Commission, 2018). This value considers both the infrastructure for the total RDC centre, as well as specific infrastructure for cold storage. Electricity, heat, and water use is also based on PEFCR guidance.

2.1.7.5 Infrastructure LCIs

This section provides LCI data and information about the relevant secondary databases used to model various infrastructure considered throughout this study, including: polytunnel and glasshouse infrastructure; cold storage facilities on farms; and distribution centres.

2.1.7.5.1 Polytunnel & glasshouse infrastructure

Table 40 provides the lifecycle inventories of different polytunnels and glasshouses used by case study farms and nurseries; these have been indicated in the relevant seedling and cultivation LCIs for kale and tomato farms, respectively, in Section 2.1.7.3 and 2.1.7.4. Each labelled polytunnel or glasshouse corresponds to a different model from a different supplier, with listed material amounts provided directly by suppliers. In cases where the supplier was not able to be contacted, or could not provide a material list, then amounts have been estimated based on similar models used on other farms. Lifetimes of materials have been estimated based on information from suppliers in the first instance, and then from secondary databases. For polytunnel plastic, the lifetime is provided by the farmer, and thus this lifetime varies based on the individual farm. The average and range of lifetimes given by farmers in the U.S. and UK has been provided.

Materials and production processes have then been modelled in SimaPro using the ecoinvent v.3.7.1 library. Additionally, for large-scale glasshouses, material amounts were based on those used in the AGRIBALYSE v.3.0 process, as information was not provided by relevant suppliers. However, this inventory has not been reproduced here, in accordance with copyright standards. Lifecycle inventories of all other polytunnels and glasshouses used are provided in Table 40. Transport burdens of the polytunnel parts from the suppliers to the farms have been modelled, but are different for each farm considered, and thus are not listed here.

Table 40 – *Polytunnel and glasshouse lifecycle inventories*

Polytunnels			**US-P-1**	**US-P-2**	**US-P-3**	**US-P-4**	**US-P-5**	**UK-P-1**	**UK-P-2**	**UK-P-3**	**UK-G-1**
Description			Polytunnel cold frame	Greenhouse (plastic)	Polytunnel quinset	Polytunnel W-truss	Polytunnel single span	Polytunnel single span	Polytunnel single span	Polytunnel single span	Hobby glasshouse
Model polytunnel size (m x m)			6 x 29	10.7 x 91.4	4 x 30.5	9 x 18	6.7 x 11	7.3 x 19.7	7.9 x 20.1	7.3 x 27.5	2.4 x 3
Material	**Process**	**Lifetime (years)**	***Material use, in kg m^{-2}***								
Steel (galvanised)	Section bar rolling	30	4.23 [a]	4.23 [a]	2.79 [a]	5.30 [a]	4.23 [a]	2.58 [a]	5.08 [a]	2.80 [a]	0
Aluminium	Section bar extrusion	25	0.363 [d]	0.710 [b,c,d]	0.321 [b,c,d,h]	0.225 [d]	0.363 [d]	0.122 [h]	0	0	13.2 [i]
Wood	Sawnwood	20	3.05 [b,c]	0.64 [b,c]	0	3.87 [b,c]	0	0.857 [b,c]	2.89 [b,c e, h]	0.834 [b,c]	0
PE	Extrusion, film	US: 6 (5-8) UK: 12 (6-18)	0.613 [f]	0.613 [f]	0.408 [f]	0.410 [f]	0.289 [f]	0.453 [f]	0.470 [f]	0.322 [f]	0
PE	Weaving, fibre	6	0.182 [g]	0.182 [g]	0	0.085 [g]	0.182 [g]	0	0	0	0
PC	Calendering sheets	15	0	0.140 [c]	0.418 [c]	0	1.16 [c]	0	0	0	0
Felt	Nonwoven, spunbond	5.5	0.051 [g]	0.051 [g]	0	0	0	0	0	0	0
Glass (3 mm)	Flat glass	35	0	0	0	0	0	0	0	0	34.5 [j]

Main uses of materials: [a] Polytunnel hoops, [b] Side-walls, [c] End-walls, [d] Connection brackets / locks, [e] Base, [f] Plastic sheet cover, [g] Roll-up curtain, [h] Door, [i] Frame, [j] Sides

[k] The lifetime of polytunnel plastic is provided by the farmer, and thus varies based on the individual farm. The average lifetime for U.S. and UK case study farms is provided with the range of lifetimes given within parentheses.

2.1.7.5.2 *Cold storage infrastructure*

For cold storage facilities, several models were considered, which included: cold storage rooms using air conditioning units; conventional refrigerated cold rooms; and standing refrigerators. For the first case, this is only used on U.S. organic farms. Air conditioning (A/C) units are placed within insulated rooms or shipping containers, and using ‘CoolBot’ systems, these A/C units are able to cool to lower temperatures than normally permitted. The infrastructure for these cold stores was modelled using relevant AGRIBALYSE processes for sheds with concrete floors or ecoinvent processes for shipping containers. Air conditioning (A/C) units were also modelled in addition.

This study specifically modelled a window-mounted A/C unit based on a small-scale, window-mounted 2.1 kW single duct room air conditioner with cooling capacity of 7,000 BTU per hour (weighing 28.926 kg). Most farmers in the study using A/C units for cool room used air conditioners of 12,000-15,000 BTU/hr and 1-2 kW, so this is comparable. Material use and manufacturing energy were based on an study for U.S. residential appliances (Boustani *et al.*, 2010). Water use during manufacturing was based on a German report of room A/C units (Schleicher *et al.*, 2018). The refrigerant was assumed to be R-410a, as is common in U.S. air conditioning units, and which is assumed in other studies (Almutairi *et al.*, 2015; G. Li, 2015; Schleicher *et al.*, 2018). The estimated lifetime of a room or window-mounted A/C unit is 10 years (Almutairi *et al.*, 2015; Schleicher *et al.*, 2018). The amount of refrigerant in this unit is estimated as 1.1 kg (0.77-1.6 kg range) based on literature sources (Almutairi *et al.*, 2015; Schleicher *et al.*, 2018) and supplier information. It is assumed that 50% of refrigerant is leaked over the 10-year lifetime (Shah, Debella and Ries, 2008; Almutairi *et al.*, 2015; Schleicher *et al.*, 2018). The amount of refrigerant refilled over the lifetime is estimated at 37.5% of the original amount (Schleicher *et al.*, 2018), with an uncertainty range included of up to 50% based on other studies (Shah, Debella and Ries, 2008; Almutairi *et al.*, 2015). Finally, it is assumed that, of the remaining amount of refrigerant present at end of life, 60% is emitted to air and 40% is sent to incineration as hazardous waste (Schleicher *et al.*, 2018). It is assumed that all metals are recycled at end of life, and other materials are sent to incineration (Schleicher *et al.*, 2018).

Refrigerated cool storage was modelled for packhouses on farms and for distribution based on the Product Environmental Footprint Category Rules (PEFCR) Guidance (European Commission, 2018). This dictates the materials used in a standard commercial fridge or freezer, which is assumed to be applicable to model cool room infrastructure, as well as providing information on energy consumption used for chilling products over the year. Lifetime of refrigerant systems is assumed to be 15 years. R404a is modelled as the refrigerant. A 10% annual leakage was applied, which was assumed to be replaced over the lifetime. Finally, it is estimated that of the remaining 90% of refrigerant in the system at end of life, 5% would be emitted to air and 95% would be sent to incineration as a hazardous waste material.

For farms using standing refrigerators for cold storage, which were only a couple small organic farms, this infrastructure has been modelled based on relevant WFLDB v.3.0 processes for commercial and domestic refrigerators, with certain adjustments. The process assumes that the typical mass of fridge is 0.24 kg per L of storage space. R404a is modelled as the refrigerant, assuming that 1% of total refrigerator mass is refrigerant gas. An annual estimated refrigerant leakage of 1% was applied, which is to be replaced over the lifetime. A 1% annual leakage was applied, which was assumed to be replaced over the lifetime, based on the IPCC report on refrigeration (Devotta and Sicars, 2005) and PEFCR guidance (European Commission, 2018). Of the remaining fraction of refrigerant gas (99%) that goes to end of life, 5% is assumed to be emitted to air during dismantling, and 94% is assumed to be incinerated as municipal hazardous waste, as per PEFCR guidance (European Commission, 2018). Average fridge lifetime in the WFLDB v.3.0 process is 14 years, which was assumed in this study, and is similar to the 15 years assumed in PEFCR guidance (European Commission, 2018). Maintenance (washing) was not included (it was in the WFLDB process, as relevant information on this was not provided by farmers and impacts from this were assumed to be negligible.

2.1.7.5.3 Distribution centre infrastructure

Distribution centres were modelled only for crops that passed through these before their final point of sale; this is assumed for all farms selling through supermarket retail chains, which for this study only included conventional farms. The infrastructure for the distribution centre was initially modelled based on ecoinvent v.3.7.1 building processes, and then was further updated based on PEFCR Guidance to include detailed material use for cold storage, energy use for general operation and for cold storage, and water use for general operation (European Commission, 2018). The distribution centre was assumed to be a 30,000 m^2 building of 5m high. Half of the total area is assumed to be used for storage. Where products are chilled, it is assumed that 10% of the total distribution centre area is used for cold storage (3,000 m^2), whilst 40% is used for ambient storage (12,000 m^2). Assuming storage up to 4m high, this gives a total storage capacity of 60,000 m3 of product over the year, or 3,120,000 m^3-weeks/yr. PEFCR guidance dictates that this storage capacity shall be allocated as follows:

- For ambient products: 4 times the product volume * stored 4 weeks
- For chilled products: 3 times the product volume * stored 1 week
- For frozen products: 2 times the product volume * stored 4 weeks

These guidelines were thus used to allocate distribution centre infrastructure to the crop of interest. Tomatoes are assumed to be stored at ambient temperatures in the U.S., and all kale and UK tomatoes are assumed to be stored as chilled products.

2.1.7.6 Compost LCIs

Composting is a process which encourages the decomposition organic material (such as food waste, yard trimmings, branches, leaves, sludge, or manure) over time, converting the material into an organic-rich product which can be used as a soil amendment (California Air Resources Board, 2017). In this study, compost is commonly utilised as an organic fertiliser by several organic farms both in the U.S. and UK. The burdens of the composting process are analysed using three different allocation options, as already discussed in Section 2.1.5. This section will provide further detail on the lifecycle inventory and fugitive emissions from the composting process.

2.1.7.6.1 Compost inputs (feedstocks)

Compost is produced using organic waste. The compost produced in this study is composed mainly of biowaste, which here refers to a mixture of green waste and food waste. Green waste includes plant materials such as waste from gardens and parks, whilst food waste refers to wasted food from either households or food manufacturers. For composting done on farms, manure may also be used as a compost feedstock (input).

Following the cut-off allocation method (Ekvall and Tillman, 1997; Kousemaker, Jonker and Vakis, 2021), the burdens of the primary lifecycle that produced the organic waste used to make compost is not included. For example, the lifecycle of the original food production which generated food waste, later to be used in compost, is not included. Similarly, the lifecycle of animals which may have produced manure is not included. Thus, the system boundary of the compost will begin with the transport of feedstock materials to the composting facility or farm, as employed by many other composting and agricultural LCAs (Martínez-Blanco *et al.*, 2010; White, 2012; Saer *et al.*, 2013; California Air Resources Board, 2017).

However, a distinction must here be made between composted manure and unprocessed manure. Some vegetable farms in the study apply unprocessed, fresh manure to their fields (often taken from nearby livestock farms). In this case, the upstream burdens of the manure (animal lifecycle) and any emissions produced from the manure itself (e.g., as it is sitting in a pile waiting to be spread) will not be attributed to the crop lifecycle. This is because it can be assumed that, if the farm did not spread the manure for crop cultivation, the manure would be decomposing in the field anyway. Consistent with Dalgaard and Halberg (2007) and Venkat (2012), the emissions from un-processed manure should be burdened only to the animal lifecycle. However, some farms further process manure into a compost, by mixing it with green waste and straw, piling it in heaps, and letting it decompose over time. This generates a compost product, which will be considered as a general "compost" for this study. In this case, because the manure is processed, the burdens of the composting process will be attributed (including decomposition emissions produced in the composting process) (Venkat, 2012). Burdens of the composting process will depend upon the compost production method used.

2.1.7.6.2 Compost production methods

Compost may be produced in a variety of production methods, in either open or closed systems. Open systems refer to those in which composting is generally done outdoor or in open-air, and thus gaseous emissions from the compost are generally not collected or treated (Sánchez *et al.*, 2015). In contrast, in closed systems, the composting takes place in an enclosed structure, where the gaseous emissions may be (and often are) collected and treated, such as with the use of biofilters (Sánchez *et al.*, 2015).

This LCA focuses on two types of compost production, based on the types and methods used to generate compost inputs used by farmers in this study. This includes: 1) home-made compost, made in heaps or piles, and 2) industrially-produced compost, produced at a commercial facility in open windrows.

Homemade compost refers to compost made at the private level, using inputs from the home (e.g., household food waste, pruning waste, or lawn / garden waste) (Boldrin *et al.*, 2009). In this study, the term 'homemade compost' is used to designate compost produced at the farm-scale for use on the same farm, with inputs from the farm itself or from nearby sources (e.g., manure from nearby farms may be used). This composting process is done using minimal equipment, and in this study, all farmers producing home-made compost do so outside in piles or heaps, using either three-sided wooden enclosures (bays) or just open piles.

In this study, the term "industrially-produced compost" refers to large-scale composting performed by private companies or municipal waste services, which may be sold commercially to a wide variety of farms and other enterprises. Industrially-produced compost may be produced by a wide variety of methods, including open technologies, such as windrow and static pile composting, or enclosed technologies, such as channel, aerated pile, and in-vessel composting (Boldrin *et al.*, 2009). The production methods investigated in this LCA is open windrow composting, as this is the production method used by the majority of compost suppliers for the farms in this study.

Windrow composting is an open composting technology, meaning that the composting occurs outdoors. In this case, gaseous emissions are generally released to the atmosphere without being collected or treated (Boldrin *et al.*, 2009). It is an aerobic (aerated) and thermophilic composting process (Recycled Organics Unit, 2007). Windrows themselves are constructed as long piles of the feedstock material, usually in triangular or trapezoidal shapes (Boldrin *et al.*, 2009); an example can be seen in Figure 9. These piles are usually turned at regular intervals with a tractor, front-loader, or other specialized equipment to promote aeration and microbial decomposition and maintain consistent and appropriate temperatures and moisture levels in the pile (Kong *et al.*, 2012). The composting method is fairly simple and requires a relatively low capital cost, and thus this is an extremely common method of composting around the world, used by farms, municipalities, and commercial producers alike (Recycled Organics Unit, 2007). Green wastes are usually the primary feedstock for windrow composting (as opposed to food waste) (Recycled Organics Unit, 2007).

Figure 9 *– Pictures of windrow composting at Longwood Plantation, LLC (Georgia, USA), a compost supplier for several organic farms in this study. Photos shared with company's permission.*

2.1.7.6.3 Compost inventory

The composting process may require a variety of resources and materials to produce a finished compost product from organic waste materials, which could be used as a fertiliser. In addition to the organic waste feedstock, inputs may include infrastructure, electricity, machinery, transport, and water. Outputs would include the finished compost, wastewater, and any material that was rejected from inclusion in the compost. Industrial composting processes generally require more material and resource inputs than home composting processes, which are simple, low-capital systems.

2.1.7.6.3.1 Industrial composting

The main input to the industrially-produced compost is the biowaste feedstock. This may be collected / transported from municipal waste streams (household organic waste collection), local authority green waste (e.g., park or other landscaping waste), food manufacturers, supermarkets, or even other agricultural sources. Thus, there is a transport impact associated with the receival of biowaste materials.

The biowaste then undergoes the composting process, which requires infrastructure for the composting facility, electricity, diesel, machinery (e.g., tractor or front-loader for turning), and water. Composting facility infrastructure, machinery, and diesel use have been estimated based on ecoinvent v.3.0 processes for industrial composting of biowaste (Nemecek and Kagi, 2007). This ecoinvent data has been based on open composting facilities, specifically using data from commercial windrow composting facilities in Switzerland (Nemecek and Kagi, 2007).

Table 41 provides an overview of the material flows included in the lifecycle inventory for industrial compost, excluding infrastructure and machinery inputs, which have been included in the LCA based on standard ecoinvent processes. This inventory is based on compost produced using windrow technology and biowaste or green waste as a feedstock.

The main input is the biowaste feedstock. The English survey of composting facilities reported 5,066,926 kg of total compost input to commercial facilities, which ultimately produced 2,676,811 tonnes of compost (WRAP, 2020). This gives a conversion factor from biowaste input to compost output of 52.8%, similar to the 50% factor used in ecoinvent v.3.0 industrial composting processes and listed by Bjarnadóttir *et al.* (2002). It is also concurrent with other studies, such as Amlinger, Peyr and Carsten (2008)'s biowaste windrow composting trials which yielded 47% and 55% compost per biowaste output. Thus, the 50% conversion factor will be applied across all industrial compost processes for this study, meaning 1 kg biowaste input will yield 0.5 kg compost.

This biowaste input must be transported from other businesses, farms, parks, or homes to the composting facility. Transport distance is dependent on the feedstock (e.g., municipal waste or local green waste), as well as the geography (distances generally longer in the U.S. versus European countries. An estimated one-way transport distance of 57 km was used, based on the average distances provided by several American and European studies (Martínez-Blanco *et al.*, 2010; Kong *et al.*, 2012; White, 2012; Saer *et al.*, 2013).

There are also residues that are often separated out as non-compost, either at the point of the feedstock reaching the composting facility; at another time before ethe composting process has begun; or during or after the composting process (WRAP, 2020). These will generally be sent for further processing or to waste (landfill); for this study, it will be assumed that all rejected material will be sent to waste due to limited information about further processing. Existing studies provide a wide range of rejection amounts, from 0.0019%-30% per weight of biowaste feedstock (Bjarnadóttir *et al.*, 2002; Martínez-Blanco *et al.*, 2009; Kong *et al.*, 2012; WRAP, 2020; ecoinvent Centre, 2021). The England-based report on composting facilities provided a figure of 0.9% average (WRAP, 2020). It is assumed that percentages for these English composting are lower than for other sources, since the English composting facilities mainly process green waste, whilst other studies investigated organic municipal solid waste as feedstocks (Bjarnadóttir *et al.*, 2002; Martínez-Blanco *et al.*, 2009; Kong *et al.*, 2012). Since the U.S. composting suppliers of interest in this study use similar technologies and feedstocks to the English suppliers, the average given by the England report will be used across industrial composting processes in this study.

Water is also emitted during the composting process, as a result of the water content in the biowaste materials; any watering of the compost; and rainwater that may fall on outdoor systems. It is estimated that 275 kg of water will be emitted per tonne of organic waste (Bjarnadóttir *et al.*, 2002), similar to the originally reported ecoinvent value of 225 kg per tonne biowaste (ecoinvent Centre, 2021). It is assumed that pollutants will be washed out of the compost and leached with this water emission. However, commercial / industrial composting facilities are generally designed to collect water run-off and send it for wastewater treatment (Bjarnadóttir *et al.*, 2002). Thus, for commercial composts water output will be modelled as being sent to municipal wastewater treatment facilities (Nemecek and Kagi, 2007).

Table 41 *– Lifecycle inventory for commercial, windrow composting of biowaste*

Flow type	**Unit**	**Amount (uncertainty range)**
Resource inputs		
Organic waste (biowaste)	kg	1.0
Electricity	kWh	0.049 (0.004-0.090) [b-f]
Diesel oil	kg	0.002 [b]
Feedstock transport (one-way)	km	57 (19-105) [d,e,g,h]
Water	L	0.418 (0.065-0.911) [i]
Outputs		
Finished compost	kg	0.5 [a-b]
Rejected material, sent to municipal solid waste	kg	0.009 [j]
Wastewater, sent for wastewater treatment	kg	0.275 (0.225-0.3) [a]

[a] (Bjarnadóttir *et al.*, 2002); [b] (ecoinvent Centre, 2021); [c] (Cadena et al., 2009); [d] (Martínez-Blanco *et al.*, 2010); [e] (Kong *et al.*, 2012); [f] (Avadí *et al.*, 2020); [g] (White, 2012); [h] (Saer *et al.*, 2013); [i] (Amlinger, Peyr and Cuhls, 2008); [j] (WRAP, 2020)

2.1.7.6.3.2 Home composting

Home or on-farm composting is only practiced on organic and relatively small-scale farms in this study. Feedstocks vary between farms, but potential ingredients include green waste (garden waste, grass clippings, leaves, agricultural residues, wood waste), food waste, and manure. When manure or woodchips are used as feedstock ingredients, they are generally transported from other nearby farms or from tree surgeons. Thus, this will have a transport impact. Otherwise, almost all of the feedstock will be from the farm, and thus will not have a transport impact associated with it.

Table 42 provides an overview of the material flows included in the lifecycle inventory for homemade compost, per kg of biowaste input. Unlike the industrial composting process, there are minimal input flows for home composting as this is generally done using minimal material infrastructure and machinery.

Home composting done by the farms in this study is all performed outside in open air, generally stacking the biowaste in heaps or piles and sometimes using three-sided wooden structures (bays) to hold the compost. Most farmers tend to their compost piles using hand tools; tractors may be used to move the compost onto a bed (although not usually), but in this case the tractor use will be counted for within crop cultivation processes. Thus, minimal physical infrastructure or machinery is used for the actual composting process, and so these have been assumed as negligible and are not counted within this study. There is also no "rejected material" as in the industrial process, as the farmer will select and sort out all improper material from food or other waste before composting it. Therefore, the only input other than possible transportation of feedstock material (which is included based on specific data provided by each farm) is water, for watering the compost, and land use.

Fewer LCA studies have been performed on home composting than industrial composting, although it is assumed home composting is less efficient in regards to conversion from waste to compost (ecoinvent Centre, 2021). However, available data gives the conversion from food waste to compost in a plastic bin home composter as 56% (Colón *et al.*, 2010), and from biowaste to compost in an wooden composter as 43% and 41% for two backyard composting systems (Amlinger, Peyr and Cuhls, 2008). Finally, AGRIBALYSE v.3.0 processes use a 65% conversion rate for home composting of green waste in heaps and a 43% conversion rate for home composting of biowaste in heaps and containers, based on experiments conducted in the south of France (APESA, OLENTICA and BIO Intelligence Service, 2015). Averaging these values provides an estimated 50% conversion rate as for industrial composting, so this will be utilised.

For home composting, water is also emitted as an output (as compost leachate). However, based on Bjarnadóttir et al. (2002)'s guidelines for the use of LCAs in the waste management sector, it is assumed that the amounts of pollutants are insignificant as long as the majority of the waste composted is garden waste; this is generally the case for the farms in this study. Thus, water will be counted as an emission output flow, but pollutants or other emissions within the water will not be counted, as they are regarded as a minor problem (Bjarnadóttir *et al.*, 2002).

Table 42 – Lifecycle inventory for home composting of biowaste

Flow type	Unit	Amount	Source
Resource inputs			
Organic waste (biowaste)	kg	1	(Amlinger, Peyr and Cuhls, 2008; Colón *et al.*, 2010; APESA, OLENTICA and BIO Intelligence Service, 2015)
Land	m^2a	0.0018	Estimate based on one farmer’s three-bay compost system, assuming a one-year composting time
Water	L	0.0908	(Colón *et al.*, 2010)
Feedstock transport (one-way)	km	4-6	Specific data from farmers
Outputs			
Finished compost	kg	0.5	(Amlinger, Peyr and Cuhls, 2008; Colón *et al.*, 2010; APESA, OLENTICA and BIO Intelligence Service, 2015)
Emitted water (leachate)	L	0.2	(APESA, OLENTICA and BIO Intelligence Service, 2015)

2.1.7.6.4 Decomposition emissions

The actual process of composting organic material in piles can produce gaseous emissions (often called fugitive emissions), which would not have otherwise occurred if the material was left to decompose naturally. Many of these emissions come from unintentionally created anaerobic conditions in the pile, which can produce gases such as CH_4 and N_2O. It is assumed in natural decomposition, only CO_2 would be produced, and thus the composting process itself introduces additional emissions that must be accounted for. While closed composting systems can capture and treat decomposition emissions, and this is not the case for open technologies. In this LCA, only open composting systems are used, and thus gaseous emissions from an open composting process will be considered.

Greenhouse gas emissions generated from the composting process are highly dependent on the waste type (feedstock) and the composition of the waste (i.e., C/N ratio, moisture percent, biodegradability, etc.) (Sánchez *et al.*, 2015). Additionally, the temperature and size of the compost pile, as well as the time of the composting process, can all affect emissions (Sánchez *et al.*, 2015). This is why emissions may differ for industrially-produced and home-produced compost. Most emissions are a result of sub-optimal conditions in the compost pile (e.g., high moisture, low aeration) which may result from poor management of the piles (e.g., infrequent turning) (U.S. EPA, 2006; Sánchez *et al.*, 2015). Thus, there is a degree of uncertainty with the emissions produced from decomposition of organic matter within the compost pile, which is accounted for within uncertainty ranges of the gaseous emissions discussed in this section.

Five gaseous emissions from the composting process (decomposition of organic matter) are considered in this LCA, including: CO_2 (biogenic), CH_4, N_2O, NH_3, and volatile organic compounds (VOCs). Although the composting process will generate a wide range of other gases, these five are considered as they account for 99% of total emissions (Chung, 2007;

Martínez-Blanco *et al*., 2009). These are also representative of the main gases considered in other LCAs focused on composting (Amlinger, Peyr and Cuhls, 2008; Boldrin *et al.*, 2009; Martínez-Blanco *et al.*, 2009, 2010; Andersen, Boldrin, Samuelsson, *et al.*, 2010; Colón *et al.*, 2010; Saer *et al.*, 2013; Cleveland *et al.*, 2017), as well as composting processes in LCA libraries, such as ecoinvent v.3.0 and AGRIBALYSE v.3.0 (Nemecek and Kagi, 2007; Avadí *et al.*, 2020; ecoinvent Centre, 2021).

CO_2 is emitted during aerobic decomposition of the organic matter by microorganisms (Kong *et al.*, 2012; Saer *et al.*, 2013). IPCC guidelines consider CO_2 emissions from the degradation of organic material as biogenic (IPCC, 2006a), meaning these emissions would occur naturally or would have happened during the natural decomposition process (Recycled Organics Unit, 2007). It is assumed that these emissions relate to a short-term carbon cycle that is relatively constant from year to year, so no net global warming occurs (Smith *et al.*, 2001). Thus, biogenic CO_2 emissions, while reported, do not contribute to the calculation of environmental impacts in this LCA, in accordance with IPCC guidelines (IPCC, 2006a) and in agreement with standard practice in other LCA composting studies (ICF Consulting, 2005; Recycled Organics Unit, 2007; Amlinger, Peyr and Cuhls, 2008; Boldrin *et al.*, 2009; Martínez-Blanco *et al.*, 2009; Saer *et al.*, 2013).

Methane (CH_4) is formed within anaerobic pockets of the compost. This can happen when compost piles are not turned (aerated) enough, or are watered too much (Recycled Organics Unit, 2007). Under anaerobic conditions, methanogenic bacteria can potentially liberate methane during decomposition of organic matter. Consensus has not been reached regarding CH_4 emissions in the composting process. Several sources assume that no CH_4 emissions occur during decomposition of organic material in the composting process (Smith *et al.*, 2001; ICF Consulting, 2005; U.S. EPA, 2006; Recycled Organics Unit, 2007). Indeed, the U.S. Environmental Protection Agency (EPA) suggests that even if CH_4 is generated within anaerobic pockets in the compost, once it reaches the oxygen-rich surface of the pile, it will be oxidised by microorganisms and converted to CO_2 (U.S. EPA, 2006).

However, other sources suggest that CH_4 emissions will occur even in well-aerated composting processes (Sánchez *et al.*, 2015). For example, Clemens and Cuhls (2003) reported the generation of methane emissions measured using biofilters across four municipal composting plants in Germany, and a wide range of other LCA studies have measured or reported CH_4 emissions from the composting process (Amlinger, Peyr and Cuhls, 2008; Martínez-Blanco *et al.*, 2009; Andersen, Boldrin, Samuelsson, *et al.*, 2010; Saer *et al.*, 2013; California Air Resources Board, 2017). Boldrin *et al.* (2009) suggests that a certain release of CH_4 does likely occur, although at modern, commercial composting sites this will likely be minimal due to optimisation of the composting process. Thus, in accordance with other composting LCAs, CH_4 emissions will be included, but an uncertainty range will be considered.

Nitrous oxide (N_2O) is mainly formed within anaerobic pockets of the compost, but can occur as a by-product of both nitrification and denitrification (Boldrin *et al.*, 2009). This normally happens at the end of the composting process, when the readily available C has been consumed without the stabilisation of all nitrogen (Boldrin *et al.*, 2009). N_2O emissions are generally considered to be an issue in poorly managed compost processes (IPCC, 2006a), often appearing when the temperature in the compost pile falls below 45°C (Amlinger, Peyr and Cuhls, 2008). Thus, N_2O has not been included as an emission in some studies (Smith *et al.*, 2001; U.S. EPA, 2006; Recycled Organics Unit, 2007; White, 2012). However, it has been measured in other studies investigating composting emissions in biofilters, with biowaste, green waste, and municipal solid waste feedstocks (Clemens and Cuhls, 2003; Amlinger, Peyr and Cuhls, 2008; Andersen, Boldrin, Samuelsson, *et al.*, 2010). N_2O is often reported in composting LCAs and greenhouse gas emission accounting frameworks, due to its high global warming potential (298 times the global warming impact per unit weight as CO_2), if emissions do occur (Boldrin *et al.*, 2009; Saer *et al.*, 2013; California Air Resources Board, 2017). Thus, N_2O emissions are included in this LCA, but an uncertainty range will be considered.

Ammonia (NH_3) may also be generated during the composting process, although there is limited consensus on this. Ammonia emissions are influenced by the aeration of the compost pile, the C/N ratio of the feedstock material, and temperature (Sánchez *et al.*, 2015). IPCC 2006 guidelines do not recognise NH_3 emissions from the composting process (IPCC, 2006a), although other studies report NH_3 when temperatures rise above 40-50°C in the compost pile (Amlinger, Peyr and Cuhls, 2008; Andersen, Boldrin, Samuelsson, *et al.*, 2010). Studies measuring emissions from municipal composting facilities have also reported NH_3 emissions in biofilters (Clemens and Cuhls, 2003; Colón *et al.*, 2009). Sánchez *et al.* (2015) highlights the importance of evaluating NH_3 as a potential emission from composting within environmental impact assessments, as it can cause acid rain. Thus, in accordance with other composting LCAs, NH_3 emissions will be included, but an uncertainty range will be considered.

Volatile organic compounds (VOCs) are organic compounds with relatively high vapour pressures, generally defined as organic compounds with boiling points less than 80°C (Komilis, Ham and Park, 2004). VOCs can include hazardous compounds that can be potential air pollutants, and they can also contribute to global warming, stratospheric ozone depletion, and tropospheric ozone formation (Komilis, Ham and Park, 2004). The main VOCs produced during the composting process include ketones, terpenes, alkanes, aromatic hydrocarbons, sulphides, organic acids, and alcohols (Sánchez *et al.*, 2015), although this is highly dependent on the feedstock material (Komilis, Ham and Park, 2004). Aeration of the compost pile also influences VOC emissions (Sánchez *et al.*, 2015). Although hazardous VOCs have been measured as being generated from the composting process (Bjarnadóttir *et al.*, 2002; Komilis, Ham and Park, 2004; Colón *et al.*, 2009), they are less commonly reported as an emission within composting LCAs. However, VOCs they have been included as an emission within the AGRIBALYSE v.3.0 compost processes (Avadí *et al.*, 2020). Due to their influence on key environmental impact categories, VOC emissions have been included in this LCA.

As noted, CH_4, NH_3, N_2O, and VOC emissions from composting are not included among all LCA studies, as many studies presume ideal conditions are maintained in the compost pile and that these emissions do not occur. Thus, in addition to analysing uncertainty ranges for each emission, a sensitivity analysis will be performed in which these emissions are not counted at all (set to 0). It should be noted that these emissions only contribute to global warming, stratospheric ozone depletion, fine particular matter formation, and terrestrial acidification impact categories within the ReCiPe 2016 midpoint impact assessment method.

An additional consideration is that the majority of these emissions, if they do occur, would be greatly reduced or avoided with the use ofenclosed composting technologies (Boldrin *et al.*, 2009; Colón *et al.*, 2009). An example is in-vessel composting, which is utilised more in the UK for treating municipal food waste as opposed to green waste (WRAP, 2020). Enclosed technologies allow for gases to be filtered and treated. However, open windrow composting is still the predominant method used by the compost suppliers relevant to this study, so these emissions will be considered without filtering.

2.1.7.6.4.1 Industrial composting

Table 43 provides the CO_2 (biogenic), CH_4, NH_3, N_2O, and VOC emissions to air, in g kg^{-1} finished compost (wet weight), produced during the industrial composting process. These values are based on other studies that specifically assess emissions from open windrow technologies (Bjarnadóttir *et al.*, 2002; Komilis and Ham, 2004; Amlinger, Peyr and Cuhls, 2008; Boldrin *et al.*, 2009; Andersen, Boldrin, Samuelsson, *et al.*, 2010; ADEME, 2012; Saer *et al.*, 2013). The included feedstocks vary between green waste, biowaste, and organic municipal solid waste. It should be noted that the compost suppliers in this study likely use primarily green waste, as this is the most common input for windrow composting. Composting time for the included studies was anywhere from 4-56 weeks. Finally, these studies took place in the U.S. and across several different European countries. Uncertainty ranges based on the minimum and maximum values included in these studies have been used to account for variations in compost production time and practices, technology, and feedstock.

Table 43 *– Emissions to air from windrow composting process (g kg^{-1} finished compost)*

Emission to air	Average value	Minimum value	Maximum value	Sources*
CO_2 (biogenic)	219	0.27	590	a-g
CH_4	1.29	0.06	35.5	b-h
NH_3	1.57	0.03	50.8	a,b,d-i
N_2O	0.38	0.0009	3.20	b-h
VOCs	1.82	0.13	11.1	f,g,i

*Sources: [a] (Komilis and Ham, 2004); [b] (Amlinger, Peyr and Cuhls, 2008); [c] (Boldrin *et al.*, 2009); [d] (Andersen, Boldrin, Samuelsson, *et al.*, 2010); [e] (ecoinvent Centre, 2021); [f] (ADEME, 2012); [g] (Avadí *et al.*, 2020); [h] (Saer *et al.*, 2013); [i] (Bjarnadóttir *et al.*, 2002)

2.1.7.6.4.2 Home Composting

Table 44 provides the biogenic CO_2, CH_4, NH_3, N_2O, and VOC emissions to air, in g kg^{-1} finished compost (wet weight), produced during home composting processes. These values are based on studies that specifically assess emissions from home composting of biowaste (green waste and food waste), either within plastic or wooden containers (Bjarnadóttir *et al.*, 2002; Amlinger, Peyr and Cuhls, 2008; Andersen, Boldrin, Christensen, *et al.*, 2010; Colón *et al.*, 2010; Martínez-Blanco *et al.*, 2010; APESA, OLENTICA and BIO Intelligence Service, 2015), or as outdoor piles (APESA, OLENTICA and BIO Intelligence Service, 2015). Composting times within these studies ranged from 12-52 weeks. This literature data is used to approximate the emissions produced during small-scale, on-farm composting in this study. Most farms within this study utilised outdoor, open-air composting systems (in contrast to enclosed systems in plastic bins); thus, there is a degree of uncertainty between the literature values reported and the emissions that might actually be produced on farms in this study. Thus, uncertainty ranges based on the minimum and maximum values included in these studies have been used to account for variations in compost production time, practices, and feedstock.

Table 44 *– Emissions to air from home composting process (g kg^{-1} finished compost)*

Emission to air	Average value	Minimum value	Maximum value	Sources*
CO_2 (biogenic)	335	158	657	a-c
CH_4	1.65	0.08	5.08	a-e
NH_3	0.56	0.03	1.5	a-f
N_2O	0.55	0.08	1.21	a, c-f
VOCs	6.14	0.03	23.0	c-e

*Sources: [a] (Amlinger, Peyr and Cuhls, 2008); [b] (Andersen, Boldrin, Christensen, *et al.*, 2010); [c] (APESA, OLENTICA and BIO Intelligence Service, 2015); [d] (Colón *et al.*, 2010); [e] (Martínez-Blanco *et al.*, 2010); [f] (Bjarnadóttir *et al.*, 2002)

2.1.7.6.5 Carbon sequestration from compost

Soil organic matter contains different fractions, or pools, of carbon, which are referred to as: readily degradable (active / labile pool), slowly degradable (slow pool), and stable (passive pool). Degradable organic matter will be oxidised under aerobic conditions by micro-organisms (a process encouraged in composting), releasing CO_2 to the atmosphere (Boldrin *et al.*, 2009; Kong *et al.*, 2012). Slowly degradable organic matter generally contains resistant plant material, such as with a high lignin content, which will likely reside in the soil over a 15-100 year time frame (U.S. EPA, 2006). On the other hand, stable organic matter may not degrade or return to the atmosphere for 100-1,000 years (Smith *et al.*, 2001; Boldrin *et al.*, 2009). Thus, carbon sequestration refers to the removal of carbon from the atmosphere and its accumulation or storage in a stable form, which essentially removes this carbon from the natural carbon cycle (Recycled Organics Unit, 2007; Kong *et al.*, 2012).

Soil carbon content can be increased through net inputs of carbon, such as by applying organic material like compost (Recycled Organics Unit, 2007). Long-term storage of carbon (sequestration) is driven by the increase of the microbially stable humus content, since most soil carbon resides in rapidly cycling, labile carbon pools (Recycled Organics Unit, 2007). Humus is a generally stable form of organic matter composed of long-chain carbon compounds. In particular, humified organic carbon, humic acid, and humin represent the most persistent pool of soil organic carbon, often residing in soils for several hundred years (Recycled Organics Unit, 2007).

Compost contains all three fractions of organic matter. Although the composting process itself encourages organic matter decomposition under aerobic conditions, it is estimated that 90% of the carbon in compost can be classified as residing in either slow or passive pools (U.S. EPA, 2006). Boldrin *et al.* (2009)'s review of the literature further suggests that a much smaller proportion of this – anywhere from 2-14% of the original C content in compost – will still be bound to the soil 100 years after compost application. Linzner and Mostbauer (2005) suggest that 5-30% of C in well-stabilised compost will be mineralised within the first two to three years.

Thus, applying compost to soils can lead to carbon sequestration. This is dependent on a wide range of biological, chemical, and management factors, such as the agricultural activities of the farm, application rate, climate, soil type, and even compost feedstock (U.S. EPA, 2006). Compost application can increase carbon sequestration through several mechanisms. For one, the addition of compost in can potentially shift the soil carbon balance to a net carbon input, allowing for some compost carbon to be retained in the soil (Recycled Organics Unit, 2007). This effect decreases with time after the first compost application, even if compost is consistently applied each year (U.S. EPA, 2006); thus, this effect is more applicable for shorter-term carbon models (10-25 years), although will likely still produce a carbon storage effect over the longer term.

Perhaps more relevant for longer-term models (like this LCA, which examines global warming potential over 100 years) is the ability of compost to produce stable carbon compounds, including humic substances and aggregates, which can be stored in the soil over longer periods of time because they are more resistant to microbial decomposition (ICF Consulting, 2005; U.S. EPA, 2006; Recycled Organics Unit, 2007). There is some evidence to suggest that the composting process produces a great proportion of humus from organic matter, in contrast to organic matter that is just left on the ground to decompose (U.S. EPA, 2006). This happens because the composting process creates ideal conditions for thermophilic (heat-loving) bacteria to predominate, which tend to produce a higher proportion of humic substances than other bacteria and fungi which predominate at ambient soil temperatures (U.S. EPA, 2006).

Finally, models evaluating carbon sequestration from compost have also suggested a 'fertilisation effect' and a 'multiplier effect' that could increase carbon storage. The fertilisation effect refers to the fact that the nitrogen in compost can stimulate high productivity of plants or crops growing in that soil, thus generating more crop residues. If returned to the soil, these residues would increase organic matter inputs and thus potentially carbon content. The multiplier effect refers to how the application of compost changes the dynamics of the carbon cycling within the soil, increasing the ability of carbon to be retained from other non-compost sources (perhaps through the formation of more stable aggregates).

However, this effect is not normally captured within soil carbon sequestration models, such as that used within the U.S. EPA's compost carbon storage assessment (U.S. EPA, 2006).

Further, indirect carbon sequestration (in terms of avoided greenhouse gas emissions) can also occur due to the additional benefits that the application of compost provides. These include the reduction of water use, by increasing water retention of the soil, and reduction of soil erosion (Kong *et al.*, 2012; Saer *et al.*, 2013; California Air Resources Board, 2017). Reduced water use occurs as compost application can increase soil porosity, creating more surface area to which water can bind (California Air Resources Board, 2017). This leads to higher water retention rates, potentially reducing amount of irrigation water needed and also decreasing soil erosion as water can be retained in the soil. Other benefits of compost use, such as increased soil microbiology and provision of micronutrients, may also result in carbon emission savings, but have not been fully quantified (California Air Resources Board, 2017).

Whether or not carbon sequestration from the application of compost should be modelled within LCAs is another area of contention (Saer *et al.*, 2013). It is difficult to estimate C sequestration, especially over long times (e.g., 100 years), because of the interaction of various carbon pools as well as local factors such as soil type and climatic conditions (Smith *et al.*, 2001). While several LCA studies do include C sequestration benefits, both direct and indirect, from compost use (U.S. EPA, 2006; Recycled Organics Unit, 2007; Blengini, 2008; White, 2012; Saer *et al.*, 2013), many also do not due to the variability of data and assumptions for C sequestration from compost (Martínez-Blanco *et al.*, 2010; Cleveland *et al.*, 2017).

Carbon sequestration from compost application is included in this study within the sensitivity analysis, but not within the main dataset. Since the impact assessment method chosen for this study assesses global warming potential over the 100-year time frame, only carbon storage values over this period were considered. This therefore includes values which relate to the fraction of organic carbon that will be stored in stable (passive) pools (Recycled Organics Unit, 2007). It is important to ensure the correct time-frame reported for carbon sequestration figures in studies, as some commonly used figures are only based on the 10-year (U.S. EPA, 2006) or 50-year (Linzner and Mostbauer, 2005) time frame, even though they are used in studies evaluating global warming potentials over 100 years (Blengini, 2008; White, 2012). For example, Saer *et al.* (2013) averages a wide range of figures reported for 30, 50, and 100 year time frames to generate a CO_2 emission saving from carbon storage, soil erosion, and reduced water use, thus reducing the reliability of these figures.

Carbon sequestration in this study will be considered in terms of direct carbon storage in soil after 100 years. The average carbon sequestration factor used is 0.101 kg CO_2-eq kg^{-1} compost (wet weight). This is based on the average of the range provided by Boldrin *et al.* (2009), which considers several literature sources and provides a range of 2-14% of carbon still bound in soil after 100 years and 5.6-38.6% C in compost. This provides the uncertainty range used as 0.004-0.198 kg CO_2-eq kg^{-1} compost. The average value used in this study is on the same order of magnitude as values reported and used by Bruun *et al.* (2006) (0.41 kg CO_2-eq kg^{-1} compost) and the U.S. EPA (0.26 CO_2-eq kg^{-1} compost), the latter of which is specifically based on carbon storage from humus accumulation only (U.S. EPA, 2006; Recycled Organics Unit, 2007). However, the figure used in this study is higher than the estimate of 0.054 kg CO_2-eq kg^{-1} compost used by Smith *et al.* (2001), based on the estimate that 8.2% C from compost will persist in the soil after 100 years.

Indirect carbon sequestration through additional benefits of compost application (e.g., increased water retention and reduced soil erosion) are included within a 'maximum' C sequestration value in an additional sensitivity analysis. However, this is considered only for the purposes of critique and comparisons with other studies. It should be noted that this value would not be actually implemented in this study, due to an uncertainty around how additional benefits are quantified and also because of the lack of data on benefits over the 100-year time frame; indeed, figures used by the California Air Resources Board estimate this over only 30 years (California Air Resources Board, 2017). Thus, the maximum carbon sequestration value will be that used by Saer *et al.* (2013), equivalent to 0.675 kg CO_2-eq (kg compost)$^{-1}$. This takes into account direct carbon storage benefits and indirect carbon benefits from increased soil water retention and reduced soil erosion.

The average and maximum carbon sequestration values for compost will be included within the sensitivity analysis whilst still including emissions from the composting process. Although the California Air Resources Board considers greenhouse gas reductions for compost from avoided burdens of landfilling, fertiliser use, and herbicide use, as well as indirect benefits of compost applications (savings from reduced soil erosion) (California Air Resources Board, 2017), this organisation specifies that benefits of direct carbon storage from C in compost should not be accounted for when subtracting landfilling burdens, as some C storage does occur within landfilling (Kong *et al.*, 2012). Further, if considering the benefits of compost application on farms, this points to the assumption that compost does have value and is not simply a by-product of the waste management process. Thus, direct carbon sequestration will be considered as a sensitivity analysis in Scenario 1 so that burdens from the composting process are still included. This will also ease in the simplicity of analysing different methods of accounting for compost burdens and benefits. Additionally, the benefits of carbon sequestration only affect the global warming impact category within this LCA, so only this category will be reported.

Carbon sequestration benefits will only be attributed to composted products in this LCA, as it is the composting process itself that is supposed to generate C storage in stable pools over the long-term (U.S. EPA, 2006). This will include any composted food waste, green waste and manure (when manure is used as a component of a home-made compost on the farm). Carbon storage benefits will *not* be applied to the spreading of un-composted, fresh manure, since the full burdens of manure storage and decomposition have not been accounted for.

2.1.8 Modelling carbon and solar energy uptake by crops

Carbon uptake in crops is modelled as per ecoinvent v.3.0 Guidelines (Nemecek and Kagi, 2007; Nemecek and Schnetzer, 2011); the same model is also used within the World Food LCA Database (WFLDB) (Nemecek *et al.*, 2015). Crops take up carbon in the form of CO_2, which is fixed in the biomass. In this model, it is assumed that the carbon in the crop's biomass is taken completely from air. The carbon uptake is modelled only for crops that are harvested and taken off the field. Any carbon within crop residues that remain on the field (including cover crops), or are composted, is not included as these residues will decompose in the ground / compost within a few years, and the carbon will thus be released.

Since values for total carbon (% C) in tomato and kale crops could not be found, total carbon is instead estimated by assuming the following C-contents of the crop's biomass components, as delineated by ecoinvent v.3.0 Guidelines: 53% C in proteins, 44% C in carbohydrates (including fibres), 75% C in fats, and 0% C in ash. CO_2 uptake for tomatoes and kale has been estimated using this calculation, as shown in Equation 2.

$$CU = \left[\left(CP_{crop} * 0.53\right) + \left(CF_{crop} * 0.75\right) + \left(Cb_{crop} * 0.44\right)\right] * \left(\frac{44}{12}\right)$$

Equation 2 – CO_2 uptake in harvested crops

Where,

- CU = the CO_2 uptake in the crop, in kg CO_2 (kg crop fresh matter)$^{-1}$
- CP_{crop} = the crude protein content per crop fresh matter (%)
- CF_{crop} = the crude fat content per crop fresh matter (%)
- Cb_{crop} = the total carbohydrate content per crop fresh matter (%), including fibre content.
- The constant values of 0.53, 0.75, and 0.44 represent conversion factors for the amount of carbon (g C) per g of protein, fats, and carbohydrates, respectively.
- $\left(\frac{44}{12}\right)$= a stoichiometric factor used to convert C to CO_2

Solar energy in biomass is also modelled as a resource input, as per ecoinvent v.3.0 guidelines (Nemecek and Kagi, 2007; Nemecek and Schnetzer, 2011). This is estimated from the energy content of the biomass (gross calorific value), which corresponds to the upper heating value of the dry biomass. Similarly for carbon uptake, this is only modelled for harvested crops. Equation 3 is used to estimate the energy content of the biomass based on biomass components; this equation has been taken from ecoinvent v.3.0 agricultural processes based on Diepenbrock (2012) and expert opinion.

$$GE = \left(CP_{crop} * 23.04\right) + \left(CF_{crop} * 38.96\right) + \left(CFb_{crop} * 17.95\right) + \left(\left(Cb_{crop} - CFb_{crop}\right) * 17.17\right)$$

Equation 3 – Gross energy in biomass of harvested crops

Where,

- GE = gross energy, the solar energy taken up in the crop's biomass, in MJ (kg crop fresh matter)$^{-1}$
- CP_{crop} = the crude protein content per crop fresh matter (%)
- CF_{crop} = the crude fat content per crop fresh matter (%)
- CFb_{crop} = the crude fibre content per crop fresh matter (%)
- Cb_{crop} = the total carbohydrate content per crop fresh matter (%), which includes fibre. Subtracting the crude fibre content CFb_{crop} from this value gives the remaining carbohydrates (available carbohydrates).
- The constant values of 23.04, 38.96, 17.95, and 17.17 are all in units of MJ per kg protein, fat, fibre, and carbohydrate, respectively; these factors are thus used to convert these biomass components to energy values.

Table 45 provides details of the values used to calculate the CO_2 uptake and gross energy in tomato and kale crops, as well as the final calculated values. The values calculated here for CO_2 uptake and gross energy in tomatoes and kale were similar (same order of magnitude) to values used in ecoinvent v.3.0 agricultural processes for tomatoes and cabbage (cabbage used to approximate for kale, since there are no kale processes). These ecoinvent values per kg tomato were 0.081-0.10 kg CO_2 for carbon uptake and 0.903-0.936 MJ for gross energy, dependent on production types and geography; per kg white cabbage, these values were 0.234 kg CO_2 and 2.72 MJ (ecoinvent Centre, 2021).

Table 45 *– Biomass components, CO_2 uptake, and gross energy in tomatoes and kale*

Property	Abbreviation	Unit	Tomato amount (range)	Kale amount (range)	Sources
Crude protein	CP_{crop}	g (100 g fresh crop)$^{-1}$	0.80 (0.5-1.1)	3.50 (1.60-5.90)	(USDA, 2019c, 2019b; Public Health England, 2021)
Crude fat	CF_{crop}	g (100 g fresh crop)$^{-1}$	0.30 (0.07-0.8)	1.50 (1.43-1.60)	(USDA, 2019c, 2019b; Public Health England, 2021)
Crude fibre	CFb_{crop}	g (100 g fresh crop)$^{-1}$	1.20 (0.7-2.0)	4.10 (4.0-4.2)	(USDA, 2019c, 2019b)
Carbohydrate	Cb_{crop}	g (100 g fresh crop)$^{-1}$	3.30 (3.00-5.96)	2.16 (1.40-3.81)	(USDA, 2019c, 2019b; Ali *et al.*, 2021; Public Health England, 2021)
CO_2 uptake	CU	kg CO_2 (kg crop fresh matter)$^{-1}$	0.077	0.145	Calculated per Equation 2
Gross energy	GE	MJ (kg crop fresh matter)$^{-1}$	0.877	1.81	Calculated per Equation 3

2.1.9 Modelling direct agricultural emissions

Emissions to air, water, and soils from fertilisers, pesticides, and crop residues have been included within this LCA. Emissions of interest were selected after assessing those that are included within the major LCI databases that model agricultural processes, such as ecoinvent v.3.0, WFLDB v.3.0, AGRIBALYSE v.3.0, and Agri-footprint v.4.0 (Nemecek, Schnetzer and Reinhard, 2016; Durlinger *et al.*, 2017; Nemecek *et al.*, 2019; Koch and Thibault, 2020), as well as those usually modelled within agricultural LCA studies.

2.1.9.1 Overview of emission models

Overall, the emissions modelled in this LCA include: direct N_2O emissions to air; indirect N_2O emissions to air, through atmospheric deposition and through leaching; CO_2 emissions to air from limestone, dolomite, and urea application; ammonia and nitrogen oxide emissions to air from nitrogen volatilisation; nitrate emissions to water from leaching; phosphorous emissions to water; and heavy metal emissions to water and soil. The calculation of these emissions differ between LCI databases, and thus models were chosen from those which were most applicable for this study and provided the most detailed estimates within the boundaries of the data available.

Sources of emissions considered are those which occur from anthropogenic activities, thus considering N and heavy metal inputs from fertiliser application, and N inputs from crop and cover crop residues left on soils. N-based emissions are driven by the inorganic nitrogen fraction in soil, which is derived from fertiliser applications and soil organic N mineralisation from microorganisms (Chen *et al.*, 2014). Organic N mineralisation can occur due to land use change or the drainage/management of organic soils (IPCC, 2019); however, these are accounted for within this LCA using ecoinvent land use change processes and so have not been calculated separately within the emission models discussed here.

Leaching emissions (of NO_3^-, P, and heavy metals) are included for all crop cultivation processes as it is assumed that all farms exist in regions where leaching / runoff occurs, consistent with major LCI databases.

Table 46 specifies which models are used for each emission calculation and provides the corresponding equation used to calculate this emission (adapted for the purpose of this LCA). Emissions are considered for seedling production; soil-based cultivation; and soilless cultivation (only relevant for UK conventional tomato production). It is assumed that soil emissions from cultivation in polytunnels behave in a similar fashion to emissions from open fields (Webb *et al.*, 2013). All emissions listed in Table 46 are modelled for soil-based cultivation; however, not all emissions are modelled for the cases of seedling production and soilless cultivation, and in some cases, different models are applied for soilless cultivation. This is specified in Table 46's "Modelled for" column and further explained in the methodology. Finally, Table 47 provides a summary of all emission factors used for the calculation of direct agricultural emissions when this is relevant.

Table 46 *– Summary of modelled agricultural emissions in LCAs*

Emission	Emitted to	Modelled for	Variable used	Equation(s)	Model used
NH_3	Air	All (cultivation of soil-based/soilless crops and seedlings) [a]	$E_{NH3,M}$ $E_{NH3,O}$ [b]	Equation 8; Equation 9	EEA 2019 Tier 2 (EMEP & EEA, 2019a, 2019b); using adapted method from (Koch and Thibault, 2020) for organic fertilisers
NO_x, modelled as NO_2	Air	All (cultivation of soil-based/soilless crops and seedlings) [a]	E_{NO2}	Equation 10	EEA 2019 Tier 1 (EMEP & EEA, 2019a),
NO_3^-, from leaching	Water	Soil-based crop cultivation [c]	$E_{NO3\text{-}S}$	Equation 11	SQCB-NO_3 model (Emmenegger, Reinhard and Zah, 2009), with sensitivity analysis including other models (see: Section 2.1.9.5.2).
	Water	Soilless crop cultivation (open systems) [d]	$E_{NO3\text{-}SL}$	Equation 18	Averaged literature values for hydroponic tomato cultivation
N_2O, direct [e]	Air	Soil-based seedling and crop cultivation	$E_{N2O,D\text{-}S}$	Equation 19	IPCC 2019 Tier 1, disaggregated (IPCC, 2019)
	Air	Soilless seedlings and crop cultivation	$E_{N2O,D\text{-}SL}$	Equation 20	Averaged literature values for soilless cultivation
N_2O, indirect from volatilisation [e]	Air	All (cultivation of soil-based/soilless crops and seedlings)	$E_{N2O,ATD}$	Equation 21	IPCC 2019 Tier 1, disaggregated (IPCC, 2019)
N_2O, indirect from leaching [e]	Air	All (cultivation of soil-based/soilless crops and seedlings)	$E_{N2O,L}$	Equation 22	IPCC 2019 Tier 1 (IPCC, 2019)
CO_2	Air	All (cultivation of soil-based/soilless crops and seedlings)	E_{CO2}	Equation 23	IPCC 2006 Tier 1 (IPCC, 2006c) [f]
P	Water	Soil-based crop cultivation	$E_{P\text{-}S}$	Equation 24	Emission factor as used in Agri-footprint v.4.0 (Durlinger *et al.*, 2017), derived from (Struijs *et al.*, 2011)
	Water	Soilless crop cultivation (open systems) [d]	$E_{P\text{-}SL}$	Equation 25	Averaged literature values for hydroponic tomato cultivation
Heavy metals	Water and soil	Soil-based crop cultivation	$M_{leach,i}$ $E_{HM\text{-}S,i}$	Equation 27; Equation 28	SALCA-heavy metal model (Freiermuth, 2006), as used in Agri-footprint v.4.0 (Durlinger *et al.*, 2017)
	Water	Soilless crop cultivation (open systems) [d]	$E_{HM\text{-}SL,i}$	Equation 29	Net balance of heavy metal inputs in fertilisers minus outputs in crop biomass
Pesticides	Air, water, and soil	All (cultivation of soil-based/soilless crops and seedlings) [a]	n/a	n/a	Agri-footprint v.4.0 Methodology (Durlinger *et al.*, 2017)

[a] Sensitivity checks are employed for these emissions for soilless crop cultivation.
[b] Two different NH_3 emission factors are modelled, designating between emissions from synthetic (mineral) fertilisers ($E_{NH3,M}$) and organic fertilisers ($E_{NH3,O}$), as these are calculated in different ways.
[c] The default model used to estimate nitrate leaching for soil-based cultivation is the de Willigen 2000 model. However, four other models are explored in a sensitivity analysis, including: the SQCB-NO_3 model (Equation 14); the Smaling 1993 model (Equation 15); the Poore-Nemecek 2018 model (Equation 16); and the general IPCC 2019 Tier 1 emission factor (Equation 17).
[d] All fertiliser-based emissions to water for soilless crop cultivation (NO_3, P, and heavy metals) are only applied to open (free draining) hydroponic systems; recirculating (closed) hydroponic systems are assumed to have 0 fertiliser-based emissions to water. This is applied to all soilless seedling production and 50% of soilless cultivation in the UK conventional tomato lifecycle.
[e] Three different N_2O emissions are accounted for, following IPCC guidance. These include: direct N_2O emissions ($E_{N2O,D}$); indirect N_2O emissions from atmospheric deposition of the volatilised fraction of N inputs ($E_{N2O,ATD}$); and indirect N_2O emissions from the fraction of leached/runoff N as nitrate ($E_{N2O,L}$).
[f] The emission factors for CO_2 emission from lime or urea were not changed in the 2019 IPCC refinement; thus, the original IPCC (2006) model is used.

Table 47 – Summary of emission factors (EFs) for direct agricultural emission modelling

Emission factor (EF)	Unit	Amount (uncertainty)	Description	Source
$EF_{NH3,M}$	kg NH_3 (kg N applied)$^{-1}$	Varies; see Table 52	NH_3 emissions from volatilisation during mineral fertiliser application; EF depends on fertiliser type and geography	EEA 2019, Tier 2 Guidance (EMEP & EEA, 2019a)
$EF_{NH3,O}$	kg NH_3-N (kg TAN applied)$^{-1}$	Varies; see Table 53	NH_3 emissions from volatilisation during organic fertiliser application (including manures); EF depends on fertiliser type	EEA 2019, Tier 2 Guidance (EMEP & EEA, 2019b)
EF_{NO2} [a]	kg NO_2 (kg N applied, after subtracting NH_3-N losses)$^{-1}$	0.04, (soil: 0.005-0.104), (soilless: 0.0041-0.082)[b]	NO_x (modelled as NO_2) emissions from volatilisation, during application of synthetic and organic fertilisers.	EEA 2019, Tier 1 Guidance (EMEP & EEA, 2019a).
$EF_{NO3\text{-}SL}$	kg NO_3-N (kg N applied)$^{-1}$	0.45 (0.18-0.59)	NO_3^- emissions from leaching for soilless crop cultivation (open hydroponic systems only)	Literature average (Massa *et al.*, 2010; Thompson *et al.*, 2013; Sanjuan-Delmás *et al.*, 2020)
$EF_{1,M}$	kg N_2O-N (kg mineral N applied)$^{-1}$	0.016 (0.013-0.-19)	Direct N_2O emissions from mineral fertiliser application for wet climates	IPCC 2019 Tier 1, disaggregated (IPCC, 2019)
$EF_{1,O}$	kg N_2O-N (kg organic N applied)$^{-1}$	0.006 (0.001-0.011)	Direct N_2O emissions from organic fertiliser and crop residue application for wet climates	IPCC 2019 Tier 1, disaggregated (IPCC, 2019)
$EF_{1\text{-}SL}$	kg N_2O-N (kg N applied)$^{-1}$	0.0087 (0.001-0.046)	Direct N_2O emissions from fertilisers application in soilless crop cultivation	Literature average (Daum and Schenk, 1996a; Yoshihara *et al.*, 2014; Llorach-Massana *et al.*, 2017; Karlowsky *et al.*, 2021)
EF_4	kg N_2O-N (kg NH_3-N and NO_2-N volatilised)$^{-1}$	0.014 (0.011-0.017)	Indirect N_2O emissions from the volatilisation of ammonia and nitrogen oxides, for wet climates.	IPCC 2019 Tier 1, disaggregated (IPCC, 2019)
EF_5	kg N_2O-N (kg N leaching/runoff)$^{-1}$	0.011 (0.00 - 0.020)	Indirect N_2O emission from nitrate leaching and runoff	IPCC 2019 Tier 1 (IPCC, 2019)
EF_{lime}	kg CO_2-C (kg limestone applied)$^{-1}$	0.12 (0.06-0.12)[c]	CO_2 emissions from limestone application	IPCC 2006 Tier 1 (IPCC, 2006c)
EF_{dol}	kg CO_2-C (kg dolomite applied)$^{-1}$	0.13 (0.065-0.13)[c]	CO_2 emissions from dolomite application	IPCC 2006 Tier 1 (IPCC, 2006c)
EF_{urea}	kg CO_2-C (kg urea applied)$^{-1}$	0.10 (0.10-0.20)[c]	CO_2 emissions from urea application	IPCC 2006 Tier 1 (IPCC, 2006c)
$EF_{P\text{-}S}$	kg P (kg P applied)$^{-1}$	0.05 (0.02-0.15)	P emissions from leaching and runoff for soil-based crop cultivation	(Struijs *et al.*, 2011), as per Agri-footprint v.4.0 (Durlinger *et al.*, 2017)
$EF_{P\text{-}SL}$	kg P (kg P applied)$^{-1}$	0.26 (0.10-0.49)	P emissions from leaching for soilless crop cultivation (open hydroponic systems only)	Literature average (Pluimers *et al.*, 2000; Sanjuan-Delmás *et al.*, 2020; Martin-Gorriz, J.F. Maestre-Valero, *et al.*, 2021)

[a] This emission factor is applied after subtracting N losses from NH_3 volatilisation, as per (Nemecek *et al.*, 2019; Koch and Thibault, 2020)

[b] The uncertainty range for the EF_{NO2} emission factor differs for soil-based and soilless cultivation. The uncertainty range for soil-based cultivation is taken from that provided in EEA 2019 guidance. The range for soilless cultivation is taken from other values used in literature (Pluimers *et al.*, 2000; Hosono *et al.*, 2006; Montero *et al.*, 2011).

[c] The uncertainty associated with emission factors EF_{lime}, EF_{dol}, and EF_{urea} are based on -50%, as per IPCC guidelines. It is estimated that emissions may be less than half of the maximum value, and the maximum value is currently used as the emission factor. Note that these IPCC 2006 emission factors were not updated in the 2019 refinement.

2.1.9.1.1 Emissions for seedling production

Emissions for seedling production are applied differently based on those produced for soil-based cultivation and soilless cultivation. The emissions for seedlings produced for each of these cultivation types will follow the relevant emissions for that cultivation. However, some emissions will be excluded.

Seedlings grown for soil-based crop cultivation are usually produced in containers with growing media (such as peat, perlite, compost, and sand) in polytunnels or greenhouses. All fertiliser emissions to water and soil are thus excluded for this seedling production. This includes NO_3^-, P, and heavy metal leaching, and heavy metal emissions to soil. This is concurrent with ecoinvent v.3.0 standards, which also do not consider these emissions for plants that are produced in glasshouses with impermeable soil cover (e.g., cement) or those that are produced in containers (Mouron *et al.*, 2017). However, all other emissions to air (except for indirect N_2O emissions from nitrate leaching) are considered for fertiliser use during seedling production.

These emissions are also excluded for seedlings produced hydroponically, although for different reasons. Emissions to soil from fertiliser use are excluded for all soilless crop processes. Emissions to water from fertiliser use are not modelled for hydroponic seedling production (for the UK conventional tomato lifecycle) because this production uses closed (recirculating) hydroponic systems, which do not produce any leachate. Emissions to air for hydroponic seedling production are modelled as for soilless cultivation.

Pesticide emissions are modelled for all processes similarly.

2.1.9.1.2 Emissions for soilless cultivation

Separate emission factors were used to estimate emissions from soilless cultivation. This is because emissions from soilless cultivation are dominated by different processes than in soil-based cultivation, thus resulting in different emission amounts. Note that these emission factors are only applied to the UK conventional tomato lifecycle, as this is the only soilless production considered in this study. Thus, where possible, calculated emission factors have been derived from literature specific to hydroponic tomato cultivation.

There is no global guidance or consensus on emission factors for soilless cultivation, as has been provided by the IPCC or EEA for soil-based cultivation (EMEP & EEA, 2019a; IPCC, 2019). Pluimers *et al.* (2000) did define specific emission factors for glasshouse tomato cultivation in rockwool, for N_2O, NO_x, NO_3, and PO_4, although these were estimated based on just a few available literature sources at the time. Thus, there is a wide variety of methodologies employed for emission modelling among LCAs on soilless cultivation. Some LCA studies do not model fertiliser application emissions at all, due to lack of consensus (Dias *et al.*, 2017); others use emission factors as derived for soil-based agriculture (Montero *et al.*, 2011; Antón *et al.*, 2012; Torrellas, Antón, López, *et al.*, 2012; Nemecek *et al.*, 2019; Maaoui, Boukchina and Hajjaji, 2021; Martin-Gorriz, J. F. Maestre-Valero, *et al.*, 2021; Arcas-Pilz *et al.*, 2022); and some aim to measure emissions from their own systems (Llorach-Massana *et al.*, 2017; Rufí-Salís *et al.*, 2020).

For this study, specific emission factors for soilless cultivation have been averaged and estimated from literature sources, where possible. These emission factors are applied for direct N_2O emissions and leached nitrate and phosphorus emissions. They are based on measured amounts of these emissions for open hydroponic systems in literature. Due to lack of available data, NH_3, NO_x, indirect N_2O, and pesticide emissions have been modelled based on emission factors used for soil-based cultivation, although indirect N_2O emissions from leaching will be dependent on values estimated for leached nitrate using soilless cultivation-specific factors. Heavy metal emissions to water are estimated using a simplified version of the calculation used for heavy metal emissions to soil for soil-based cultivation; this is based on subtracting heavy metal uptake in the crop from the heavy metal input in fertilisers.

Emissions to water (from leaching) are only considered for open hydroponic systems, which do not recirculate the drained leachate (wastewater). It is assumed that closed (recirculating) hydroponic systems produce no leachate (the only water loss occurs from evapotranspiration) and thus no leached nutrient emissions (Montero *et al.*, 2011; Martin-Gorriz, J. F. Maestre-Valero, *et al.*, 2021; Rufí-Salís *et al.*, 2021). For the UK conventional tomato lifecycle, this applies to 50% of the total hydroponic setup (i.e., 50% of systems are closed and 50% are open, based on industry communication). Heavy metal emissions to soil are not considered for hydroponic systems, in line with AGRIBALYSE v.3.0 soilless cultivation processes (ADEME, 2020; Koch and Thibault, 2020).

Specific emission factors used are discussed in more detail within the relevant emission sections.

2.1.9.1.3 Excluded emissions

The following emissions are not modelled within this study:

- Soil loss & consequent emissions from erosion (included in AGRIBALYSE v.3.0 and WFLDB v.3.0 databases, but not in Agrifootprint v.4.0).
- N_2O from the nitrogen fixation of N-fixing crops (e.g., green manures) is not included, based on IPCC 2006 and 2019 guidelines. The 2006 IPCC guidelines removed biological nitrogen fixation as a direct source of N_2O, updated from the 1996 guidelines, because of the lack of evidence of significant emissions arising from the fixation process itself (Rochette and Janzen, 2005). The IPCC 2006 and 2019 guidelines thus confer that N_2O emissions from the growth of any nitrogen-fixing should be modelled simply based on emissions that occur from their crop residues, dependent on the N content of above-ground and below-ground residues, as for other non-nitrogen fixing crops (IPCC, 2006c, 2019).
- Emissions from natural mulches provided to the farmer for free (e.g., pinestraw, leaf mulch, or wood chips), which are assumed to have decomposed naturally anyway, on or off the farm. Emissions from purchased mulch materials (e.g., bought-in hay) are counted since they are being brought into the farm boundaries and are considered a valuable product, not a waste material.
- Trace metal emissions from crop residues and cover crop residues.
- Direct emissions to soil from the input of nutrients (N, P, K, Ca, S etc.). These are not counted because it is generally assumed that the farmer is following fertiliser recommendations, and thus these nutrients will be taken up mostly by the crop; this is consistent with assumption in the ecoinvent database (Nemecek and Kagi, 2007).

2.1.9.2 Inputs to emission calculations

The modelling of emissions is based on inputs to the soil. The emission modelling for NH_3, NO_x, N_2O, and NO_3^- is applied to the nitrogen content of inputs, whilst P leaching depends on applied P_2O_5. The main nitrogen-based emissions in this project occur from N inputs of: synthetic fertilisers, organic fertilisers, and crop residues. P leaching is considered only for the P_2O_5 content of fertilisers.

Certain emissions are calculated separately for synthetic (mineral) and organic fertilisers. Thus, these must be defined. Synthetic fertilisers are mineral-based NPK fertilisers, as used by the conventional farms in this study. Organic fertilisers are taken to include any composts, compost teas, manures, or manufactured, certified organic fertilisers, as used by the organic farms in this study. It should be noted that, for composts, this section accounts for emissions only from application, which is separate from the pre-discussed emissions resulting from the composting process, as done in other agricultural LCAs (Venkat, 2012).

Before calculation of any emissions can be performed, N and P_2O_5 contents of inputs must be determined.

2.1.9.2.1 Fertilisers

NPK (nitrogen, phosphorous, and potassium) contents of mineral (synthetic) fertilisers are taken directly from farmers or suppliers, which always provide this information on fertiliser labels.

The N content of organic inputs are based either on information provided by manufacturers, for produced organic fertilisers used or for similar products; on information provided by farmers directly; or from academic literature or U.S. agricultural extension research. NPK contents for the common types of organic inputs used in this study are depicted in Table 48 (note: this is not an exhaustive list of all organic fertilisers applied, but the most common ones). For homemade mixes of inputs (e.g., composts or compost teas made on farms), NPK values are calculated based on the NPKs of each input and their relative proportions in the mix.

***Table 48** – NPK content of common organic fertilisers used in the LCA study*

Organic input	Total N (%)	P_2O_5 (%)	K_2O (%)	Source
Green waste compost (U.S.)	1.01	0.25	0.29	Primary data (supplier)
Green waste compost (UK)	0.81	0.33	0.66	Primary data (supplier)
Poultry manure	1.1	0.8	0.5	Primary data (farmer)
Cow manure	0.41	0.15	0.58	Agrifootprint v.4.0 processes (Durlinger *et al.*, 2017)
Horse manure	0.6	0.25	0.45	(Fabian and Smith-Zajaczkowski, 2019)
Horse manure with bedding	0.7	0.2	0.7	(Fabian and Smith-Zajaczkowski, 2019)
Rabbit manure	2.4	1.4	0.6	Primary data (farmer)
Dried, pelleted poultry manure (US)	5	4	3	Primary data (supplier)
Dried, pelleted poultry manure (UK)	4.5	3.2	3.1	Primary data (supplier)
Feathermeal	13	0	0	Primary data (supplier)
Fishmeal, product 1	2	4	1	Primary data (supplier)
Fishmeal, product 2	5	1	1	Primary data (supplier)
Fish / seaweed fertiliser blend	2	3	1	Primary data (supplier)
Okraseed meal	4.88	1.66	1.95	Primary data (farmer)
Alfalfa meal	3	2	2	Primary data (supplier)
Worm castings	0.8	0	0	Primary data (supplier)

2.1.9.2.2 N content of crop residues

Nitrogen-based emissions also come from crop residues. The emissions modelled here consider crop residue emissions simply based on viewing the residues as a nitrogen input to soil, and any emissions from the actual decomposition process are not considered due to lack of sufficient information and scientific consensus regarding these emissions, in line with IPCC 2019 guidelines (IPCC, 2019).

Crop residues are mainly composed of organic nitrogen (>91% of total N), but do have a small remaining soluble nitrogen fraction (Justes, Mary and Nicolardot, 2009; Lashermes *et al.*, 2022), which is available to growing plants and contributes to the soil inorganic nitrogen pool that drives the N-based emissions modelled here. The addition of crop residues to the soil can either result in net immobilisation (reduction of the inorganic soil N) or net mineralisation (increase of inorganic soil N). This is largely dependent on the C:N content of the residues; for C:N contents between 9.4-22.7, which includes most vegetable crop residues, green manure, and leguminous residues, net mineralisation generally occurs (Chen *et al.*, 2014). Thus, the mineralisation of organic nitrogen in crop residues will further contributes to the inorganic N pool (Chen *et al.*, 2014). This process occurs over time, although most rapidly at the start, when residues are returned to the soil (Justes, Mary and Nicolardot, 2009; Chen *et al.*, 2014). However, IPCC models of N_2O emissions from crop residues do not account for any delay in emissions over time, and instead consider emissions from total crop residue N during the year in which the residues are incorporated into the soil (IPCC, 2019).

Crop residues accounted for in this study include the harvest waste of the crop left in the field, as well as above-ground and below-ground crop residues. Many farmers in the study either leave these residues to decompose on the field surface or incorporate residues into the soil directly (e.g., through ploughing), so these residues will result in nitrogen-based emissions to air (direct N_2O and indirect N_2O emissions) and water (nitrate leaching). Crop residues that are exported from the field (not incorporated) are not considered in these emissions. In this case, farmers either compost these residues, and thus relevant emissions are accounted for within the composting process, or they may be fed to animals, in which case emissions from decomposition or incorporation do not occur.

Nitrogen in residues from cover crops or green manures, either grown directly preceding the crop of interest or within a long-term (5-6 year) rotation, are also considered. N amounts are allocated to the crop of interest in the same fashion that land use and inputs for the cover crop process are allocated. All farmers in this study who utilised cover crops to improve soil fertility or overall soil health did not harvest any of the cover crops; thus, all cover crops were incorporated directly into the soil, and N-based emissions are calculated as for other crop residues.

To calculate resulting N-based emissions to air and water from crop residues, the amount of N in crop residues (F_{CR}) must first be calculated. This has been done based on IPCC methodology (IPCC, 2006c, 2019)

2.1.9.2.2.1 N content of cover crop residues

The amount of N in cover crops that are incorporated into the soil are calculated by adapting standard IPCC equations (IPCC, 2006c) for this project. The IPCC equations use constant factors to estimate crop residues based on the amount of crop harvested; the ratio of above-ground residues to harvested crop ($R_{AG(T)}$); and the ratio of below-ground residues (i.e., root biomass) to harvested crop *or* the ratio of below-ground residues to above-ground biomass (RG_{BIO}), as used here. For the cover crops in this project, this equation must be adapted because there is no harvested crop - all above-ground biomass is incorporated into the soil. Therefore, Equation 4 is used as the adapted version. The output of this equation provides the total N from cover crop residues (F_{CC}) that are incorporated to the soil within the total time period that the cover crop is grown (accounted for within the yields); this amount is then later allocated to the crop of interest within the LCA.

$$F_{CC} = \left[\left(DMyield_{(T)} * Area_{(T)} \right) * \left(\left(R_{AG(T)} * N_{AG(T)} \right) + \left(RG_{BG-BIO(T)} * N_{BG(T)} \right) \right) \right]$$

Equation 4 - N in cover crop residues, adapted from IPCC, 2006c;2019

Where,

- F_{CC} = the amount of N in cover crop residues (above- and below-ground), returned to soils annually in kg N year^{-1}.
- $DMyield_{(T)}$ = the dry matter (DM) yield of the cover crop *T*, as kg dry matter ha$^{-1,}$ provided in Table 49.

- $Area_{(T)}$ = the total area where the cover crop *T* is grown, in ha $year^{-1}$.
- $R_{AG(T)}$ = the ratio of above-ground residue dry matter to harvested yield for the crop *T* (dimensionless). For cover crops, this always equals 1, since the crop DM yield equates to all above-ground residue dry matter.
- $N_{AG(T)}$ = the N content of above-ground residues for cover crop *T*, in kg N (kg DM in above-ground residues)$^{-1}$, provided in Table 50.
- $R_{BG\text{-}BIO(T)}$ = the ratio of below-ground residues to total above-ground biomass for cover crop *T* (dimensionless), provided in Table 50.
- $N_{BG(T)}$ = the N content of below-ground residues for cover crop *T*, in kg N (kg DM in below-ground residues)$^{-1}$, provided in Table 50.

The dry matter yields utilised for cover crops in this project are provided in Table 49. These yields are listed as total above-ground dry matter produced per area per year. The management assumptions associated with these yields are also provided; this relates to the number of cuts taken on the cover crop over the year to achieve the listed yield. This information was then used to apply the yields for different farmers' systems, as some farmers did not take as many cuts as assumed in the yields provided. Finally, the table also lists whether the particular cover crop was used by UK farmers in the study, U.S. farmers in the study, or both (UK & U.S.).

For the most common cover crops used in the UK, yield values were taken from those provided Cotswold Seeds, a major cover crop and herbal ley seed provider in the UK; these were gleaned from their catalogue and website in 2022 (https://www.cotswoldseeds.com/). Cotswold Seeds provides the maximum dry matter yield achieved, so estimations for farmers in this project have been reduced from that maximum value.

For the most common cover crops used by farmers in Georgia, U.S., yield values were taken from the cover crop handbook produced by the U.S.-based Sustainable Agriculture Research and Education (SARE) program (SARE, 2007). The handbook provides a range of yield values for different crops, which were taken from published research, some with multi-cut systems. The handbook recommends that farmers' actual dry matter yield would likely be within the minimum – midpoint range that they provide; thus, for single-cut systems, dry matter yield was estimated as the average between the minimum and midpoint dry matter yield provided.

When both sources provided information on a certain cover crop, preference was given based on the location where most farmers used that cover crop (i.e., U.S. vs. UK), or both sources were taken into account and an average was produced. Finally, for only one cover crop (sunflower), neither source provided dry matter yields, so this was estimated based on UK and European literature sources (Edwards *et al.*, 1978; Ion *et al.*, 2014).

__Table 49__ – Cover crops and dry matter yields

Cover Crop	DM yield (kg ha*yr^{-1})	Management Assumptions	Location Used	Source
Red clover	6,000	2-3 cuts per year	UK & US	Cotswold Seeds
White clover	7,000	2-3 cuts per year	UK & US	Cotswold Seeds
Italian ryegrass	16,000	3 cuts per year	UK & US	Cotswold Seeds
Hairy vetch	5,500	3 cuts per year	UK & US	Cotswold Seeds
Mustard	4,500	1 cut per year (when incorporated)	UK	Cotswold Seeds
Fodder radish	2,500	1 cut per year (when incorporated)	UK	Cotswold Seeds
Wheat	4,760	1 cut per year (when incorporated)	US	(SARE, 2007)
Buckwheat	2,800	1 cut per year (when incorporated)	US	(SARE, 2007)
Oat	4,480	1 cut per year (when incorporated)	US	(SARE, 2007)
Cereal / grazing Rye	5,500	1 cut per year (when incorporated)	UK & US	(SARE, 2007) & Cotswold Seeds
Pea	4,760	1 cut per year (when incorporated)	US	(SARE, 2007)
Cowpea	3,360	1 cut per year (when incorporated)	US	(SARE, 2007)
Sunflower	16,000	1 cut per year (when incorporated)	UK & US	(Edwards *et al.*, 1978; Ion *et al.*, 2014)
Giant sorghum (Sorghum-sudan)	9,500	1 cut per year (when incorporated)	UK	(SARE, 2007)

Estimations of $N_{AG(T)}$, $R_{BG\text{-}BIO(T)}$, and $N_{BG(T)}$ for basic groups of crops are provided within the IPCC 2006/2019 guidelines; note that values were not updated in the 2019 version, but uncertainties were added (IPCC, 2006c, 2019). The factors utilised in this project have been reproduced in Table 50 for convenience. The 'IPCC crop type' column delineates the major crop type listed in the IPCC guidelines and the 'Cover crops included in this category' column lists the cover crops used by farmers in this project which relate to the IPCC category. Note that N content of cover crop residues (forages, herbal leys, etc.), as applied in Equation 4, is assumed to be the N content of the whole above-ground crop biomass.

Table 50 – *N content of above-ground and below-ground biomass in cover crops*

Crop type (as per IPCC)	**Cover crops included in this category**	**N_{AG} (kg N kg DM^{-1})** [a]	**N_{BG} (kg N kg DM^{-1})** [b]	**$R_{BG\text{-}BIO}$** [c]
Grains	Wheat, buckwheat, cereal/grazing rye, oats	0.006 (± 75%) [d]	0.009 (± 75%)[d]	0.22 (± 16%)
Beans & pulses	Peas, cowpeas	0.008 (± 75%) [d]	0.008 (± 75%)[d]	0.19 (± 45%)
Grass-clover mixes	Any mixtures of only grasses and clovers	0.025 (± 75%) [d]	0.016 (± 75%)[d]	0.80 (± 50%)
N-fixing forages	Any herbal leys or mixes which include N-fixing crops and other species, and are not strictly grass-clover mixes	0.027 (± 75%) [d]	0.022 (± 75%)[d]	0.40 (± 50%)
Non-N-fixing forages	Any herbal leys or mixes which include grasses and other non-N-fixing forage crops (such as vetch)	0.015 (± 75%) [d]	0.012 (± 75%)[d]	0.54 (± 50%)

[a] N content of above-ground residues; [b] N content of below-ground residues; [c] ratio of below-ground residues to above-ground biomass.
[d] These uncertainty values were not provided in studies, but rather estimated as expert judgement from the IPCC (IPCC, 2019).

2.1.9.2.2.2 N content of main crop residues

Table 51 displays the factors used to estimate above-ground and below-ground crop residue N for tomatoes and kale.

Table 51 *- Tomato and kale crop residue biomass and nitrogen content*

Factor	Abbreviation	Unit	Tomato amount (range)	Kale amount (range)	Description	Source
Plant Above-Ground Residue: Harvested Crop	$AG_{crop}:H_{crop}$	n/a	0.3	2.17 (1.66-3.12)	Ratio of above-ground (AG) plant residue weight to harvested crop weight, in terms of fresh weight. For tomato plants, AG residue includes stems and leaves but not fruit; for kale, this includes stalk and non-edible leaves.	Tomatoes: (Di Blasi, Tanzi and Lanzetta, 1997), in Italy Kale: (Groenbaek *et al.*, 2014), in Denmark
Stalk : Harvested Crop	$S_{crop}:H_{crop}$	n/a	n/a	1.31 (1.0-1.89)	Ratio of the stalk weight to harvested kale crop weight (edible leaves only), in fresh weight.	Kale: (Groenbaek *et al.*, 2014)
Below-ground DM : Fruit DM	$BG_{DM(crop)}$: $F_{DM(crop)}$	n/a	0.069	n/a	Ratio of tomato roots (below-ground biomass), to total tomato fruit weight (harvested and unharvested fruit), in dry matter.	Tomato: (Ronga *et al.*, 2019), based on Italian processing tomatoes
Below-ground DM: Above-ground DM	$BG_{DM(crop)}$: $AG_{DM(crop)}$	n/a	n/a	0.08 (0.03-0.16)	Ratio of roots (below-ground biomass), to total AG biomass of kale, including harvestable and non-harvestable leaves, stems, and stalk, in dry matter.	Kale: (Urlić *et al.*, 2016), in Croatia
Crop Dry Matter	DM_{Crop}	%	8.2 (3.83-31.97)	14.95 (11.6-21.8)	Dry matter percent of main crop weight (tomato fruit and kale leaves).	Tomato: (Ali *et al.*, 2021), literature review Kale: Averaged literature values (Lefsrud *et al.*, 2008; Public Health England, 2021)
Plant Above-Ground Residue Dry Matter	$DM_{AG(crop)}$	%	15.2 (10-20.4)	13.43 (10.0-20.0)	Dry matter percent of above-ground residue weight (waste after harvest).	Tomato: (Maniadakis *et al.*, 2004), in Crete Kale average: (Elsa and Desmond, 2021), in U.S. Kale range: (Di Blasi, Tanzi and Lanzetta, 1997), in Italy, based on range given for cabbage residues.
Crop N	N_{crop}	%	1.73 (1.48-3.0)	4.74 (2.7-5.98)	% N in the crop of interest (tomato fruit or kale leaves), per dry matter of crop.	Tomato & kale average: (Public Health England, 2021) Tomato range: (Maher, 1976; Kuntoji *et al.*, 2021) Kale range: (Groenbaek *et al.*, 2014; Yoder and Davis, 2020; Elsa and Desmond, 2021)
Above-Ground Residue N	$N_{AG(crop)}$	%	2.95	2.04 (1.86-2.64)	% N in above-ground residue (plant residue including stems, leaves, and vines, but excluding tomato fruit and harvested kale leaves)	Tomato: Averaged between two studies: (Maniadakis *et al.*, 2004), and (Kuntoji *et al.*, 2021), in India Kale: (Groenbaek *et al.*, 2014)
Stalk N	$N_{stalk(crop)}$	%	n/a	16.15 (9.70-17.7)	% N in kale stalk (stalk only)	Kale: (Groenbaek *et al.*, 2014)
Below-Ground Residue N	$N_{BG(crop)}$	%	1.2	1.2	%N in below-ground crop residue (roots)	Tomato & Kale: based on the IPCC value for non-N-fixing forage crops (Table 50) since there was no literature data available (IPCC, 2006c)

Using the values provided in Table 51, the following equations were used to estimate the amount of N in waste crop residues left to decompose in the field.

For tomatoes (*tom*), it was assumed that the figure of harvest waste (*HW*) provided by famers includes all fruit left in the field, whereas the harvest figure (*H*) includes all tomato fruit harvested from the field. Thus, the sum of the harvest and harvest waste includes all fruit produced by the tomato plant (F_{tom}). To estimate % N in tomato crop residues, Equation 5 is used, which is essentially an adapted version of Equation 4 from IPCC 2006 guidelines. The total crop residue N includes the added values of: N in harvest waste (tomato fruit left unharvested in field); N in above-ground tomato plant residues, estimated using a ratio for above-ground tomato plant biomass to harvested tomato crop; and N in below-ground residues (tomato roots), estimated using a ratio for below-ground tomato plant biomass to total tomato fruit produced.

The contribution from harvest waste is multiplied by a compost factor *CF*, which is based upon information provided by farmers; if the harvest waste is taken from the field to be composted, or is fed to animals on the farm, *CF* = 0. If the harvest waste is left in the field, *CF* = 1. This is applied in accordance with the cut-off method employed in this LCA and to avoid double-counting of emissions. If crop residues are composted, then emissions from decomposition will already be counted within emissions of the composting process; if crop residues are fed to animals, then any downstream impacts would be encompassed within the impacts of the animal's lifecycle.

$$\begin{aligned} F_{CR,tom} = \big[(&HW_{tom} * DM_{tom} * N_{tom} * \ CF_{fruit}\big) \\ &+ \big(H_{tom} * AG_{tom}{:}\,H_{tom} * DM_{AG(tom)} * N_{AG(tom)} * \ CF_{plant}\big) \\ &+ ((H_{tom} + HW_{tom}) * DM_{tom} * BG_{DM(tom)}{:}\,F_{DM(tom)} * N_{BG(tom)})\big] \end{aligned}$$

Equation 5 - N in tomato crop residues

Where,

- $F_{CR,tom}$ = the amount of N in tomato crop residues (above- and below-ground), returned to soils annually in kg N year^{-1}. This includes N in harvest waste (tomato fruit), above-ground plant residues (tomato haulm), and below-ground plant residues.
- HW_{tom} = harvest waste of tomato fruit left in the field, in [kg], as provided by farmers.
- CF_{fruit} and CF_{plant} = compost factor, for the *fruit* (waste tomato fruit) and for the *plant* (waste plant biomass, excluding fruit). *CF* = 0 if residues are composted, or used as animal feed, and = 1 if residues are incorporated into the field.
- H_{tom} = harvested crop of tomatoes taken out of the field, in [kg], as provided by farmers.
- $F_{DM(tom)}$ = total fruit dry matter [%], taken as the sum of the harvest and harvest waste, in dry matter: $(H_{tom} + HW_{tom}) * DM_{tom}$
- Values and descriptions for all other variables (DM_{tom}, N_{tom}, AG_{tom}:H_{tom}, $DM_{AG(tom)}$, $N_{AG(tom)}$, $BG_{DM(tom)}$:$F_{DM(tom)}$, and $N_{BG(tom)}$ are provided for tomatoes (tom) in Table 51 using literature values.

For the kale plant, the non-harvestable waste is less distinct since this includes non-edible leaves (tops of the stalk, as well as blemished leaves) left on the stalk. Some farmers have provided an estimate of harvest waste which includes non-edible leaves left in the field.

However, others have not considered non-edible leaves as a harvest waste, and thus have not provided any estimate of non-harvested leaves left in the field. In this case, they have assumed that they have harvested all edible leaves and thus have provided figures of 0 harvest waste. Because of these differing views and cases, two different equations have been developed to estimate N from kale crop residues in the field. Where possible, the values of harvest waste provided by farmers (primary data) have been utilised to estimate above-ground residue left in the field, in preference over literature values estimating above-ground residues (secondary data).

In the case where farmers have provided an estimate of non-edible leaves left in the field as harvest waste (HW), it is assumed that this figure is an approximate value for all non-edible leaves left in the field. Thus, the only additional above-ground residue that needs to be accounted for is the kale stalk (S). In this case, the following Equation 6 is used. Similar to Equation 5, this includes the added amounts of N in non-edible kale leaves (harvest waste); the amount of N in the kale stalk residue, estimated using the ratio of kale stalk weight to harvestable kale weight; and the amount of N in below-ground kale residues (roots), estimated by summing the total above-ground kale biomass (harvest, harvest waste, and stalk, in dry matter weights) and using the ratio of below-ground kale dry matter to total above-ground kale plant dry matter.

In the case where farmers have not provided an estimate of kale leaves left in the field, overall crop residues of non-edible leaves and stalk will be estimated using the ratio of kale above-ground plant residue to harvested crop (AG_{crop}:H_{crop}) provided from literature values in Table 51. The total crop N will thus be calculated from the added values of: total above-ground kale plant residue N, from non-edible leaves and stalks taken together; and N in below-ground residues (kale roots). The updated Equation 7 is thus used to calculate the amount of N in kale crop residues when a harvest waste figure for non-edible kale leaves has not been provided.

Once calculated, all values of F_{CR} for cover crop and main crop residues will then be utilised in within emissions equations to estimate N_2O emissions to air and nitrate emissions to water that come from these residues.

$$F_{CR,kale} = [(HW_{kale} * DM_{kale} * N_{kale} * CF_{leaves}) + \left(H_{kale} * S_{kale}{:}H_{kale} * DM_{AG(kale)} * N_{S(kale)} * CF_{stalk}\right) + (\left((H_{kale} + HW_{kale}) * DM_{kale}\right) + \left(H_{kale} * S_{kale}{:}H_{kale} * DM_{AG(kale)}\right) * * BG_{DM(kale)}{:}AG_{DM(kale)} * N_{BG(kale)})]$$

Equation 6 - N in kale crop residues, when harvest waste is provided

$$F_{CR,kale}\left[\frac{kg\ N}{yr}\right] = \left[(H_{kale} * AG_{kale}{:}H_{kale} * DM_{AG(kale)} * N_{AG(kale)} * CF\right) + (H_{kale} * AG_{kale}{:}H_{kale} * DM_{AG(kale)} * BG_{DM(kale)}{:}AG_{DM(kale)} * N_{BG(kale)})]$$

Equation 7 - N in kale crop residues, when harvest waste is not provided

Where,

- $F_{CR,kale}$ = the amount of N in kale crop residues (above- and below-ground), returned to soils annually in kg N year^{-1}. This includes N in harvest waste (kale leaves), above-ground plant residues (kale stalk), and below-ground plant residues.
- HW_{kale} = harvest waste of non-edible kale leaves left in the field, in [kg]. This figure is used when farmers have provided a non-zero estimate of kale leaves left in the field.
- H_{kale} = harvested crop of kale leaves taken out of the field, in [kg]. This figure is provided directly by farmers in the study.
- CF_{leaves} and CF_{stalk} = compost factor, for the *leaves* (waste kale leaves) and for the *stalk* (kale stalk biomass; this is set based on information provided by farmers about whether or not they compost kale leaf and stalk residues. *CF* = 0 if residues are composted, or used as animal feed, and =1 if residues are incorporated into the field.
- Total above-ground kale biomass dry matter ($AG_{DM(kale)}$) is taken as the sum of all kale leaves (harvest and harvest waste) and the stalk, in dry matter: $\left((H_{kale} + HW_{kale}) * DM_{kale}\right) + \left(H_{kale} * S_{kale}{:}H_{kale} * DM_{AG(kale)}\right)$
- Values and descriptions for all other variables (DM_{kale}, N_{kale}, S_{kale}:H_{kale}, $DM_{AG(kale)}$, $N_{S(kale)}$, $BG_{DM(kale)}$:$AG_{DM(kale)}$, $N_{BG(kale)}$, and AG_{kale}:H_{kale} are provided for kale in Table 51 using literature values.

2.1.9.3 Ammonia emission calculations

Ammonia (NH_3) emissions occur when NH_3 in solution is exposed to the atmosphere, resulting in volatilisation and later the deposition of this gas and its products onto soil and water bodies (IPCC, 2019). NH_3 volatilisation in agriculture can occur through several pathways (EMEP & EEA, 2019a). First, emissions can occur directly during fertiliser application; this only happens from fertilisers that contain N as ammonium (NH_4^+), or if the N in fertiliser is rapidly decomposed into NH_3, as for urea. Fertilisers containing only nitrate N are not direct sources of ammonia emissions. For volatilisation to occur, NH_3 must be in solution; however, for solid fertilisers, usually there is enough moisture in the air or in the soil for the fertiliser to dissolve. At the point of fertiliser application, the extent of NH_3 emissions will be dependent upon the concentration of NH_3 in solution, the temperature of the solution, the surface area exposed to the atmosphere, and the concentration gradient of NH_3 in the atmosphere (EMEP & EEA, 2019a).

Additional emissions from the crop canopy itself can occur shortly after N fertiliser application (7-10 days later) (EMEP & EEA, 2019a). This happens because of the increase in N concentration in plant leaves after fertilisation, and thus these indirect emissions from the canopy can happen after fertilisation with any N fertiliser (not just those containing ammonium). These emissions that happen just after fertilisation cannot be distinguished from direct emissions that occur during fertiliser application, and thus these emissions are usually included within those counted for fertiliser application (EMEP & EEA, 2019a). These collective NH_3 emissions are modelled within this study.

Additional NH_3 emissions can also occur later in the crop lifecycle, depending on the N content of the plant; the growth stage; any stresses, such as disease or drought; the time of day; and the ambient NH_3 concentration (EMEP & EEA, 2019a). Finally, during the senescence of standing plants and also crop residues, net NH_3 emissions can occur as proteins break down and form NH_4^+. However, there is much scientific difficulty and uncertainty in the estimation of NH_3 emissions from both standing crops and crop residues, due to limited measurements of NH_3 flux in agricultural fields across full seasons or years (EMEP & EEA, 2019a). Thus, no suitable methodology has yet been developed to estimate these emissions, and therefore they are not included in this LCA. This is consistent with methodologies from both the IPCC and EEA, which also do not estimate NH_3 emissions from standing crops or crop residues (EMEP & EEA, 2019a; IPCC, 2019).

NH_3 emissions in this LCA are modelled after EEA 2019 guidance, which provides separate emission factors and methodology for synthetic and organic fertilisers. These are detailed in the following sections.

2.1.9.3.1 Synthetic fertilisers

For synthetic fertilisers, NH_3 emissions are calculated using EEA 2019 Tier 2 emission factors (EMEP & EEA, 2019a). These emission factors are applied to all synthetic (mineral) fertilisers used in this study, including those used in soilless cultivation and seedling production.

NH_3 emissions are influenced by factors such as the ammonium content in the fertiliser, the soil pH, and the temperature, where higher ammonium content, higher soil pH and higher temperatures favour NH_3 emissions. Thus, EEA 2019 guidance has provided emission factors based upon the type of mineral fertiliser (taking into account ammonium content); the pH of the soil; and the climate in the farm of interest (EMEP & EEA, 2019a).

Emission factors are grouped based on “cool”, “temperate”, or “warm” climates, and “normal” or “high” pH. Climate is defined based on the average annual temperature (°C) of the site of interest, where <15 is “cool”, 15-25 is “temperate”, and >25 is “warm.” A pH of ≤7 is considered “normal”, while >7 is “high.” Within these definitions, all UK soil-based farms are within a cool climatic zone whilst all Georgia farms are within a temperate climatic zone. All farms that use synthetic fertilisers in Georgia and England fall within the normal pH category, although Georgia soils are generally more acidic. Soil pH for farms was verified using national soil surveys and maps (UK Soil Observatory, 2007; Emmett *et al.*, 2010; NRCS, 2021).

The EEA emission factors are also applied for soilless cultivation and seedling production, in line with methodologies employed in ecoinvent v.3.0, WHDLB v.3.0, and AGRIBALYSE v.3.0 for soilless crops. The emission factors used for UK soilless cultivation are those defined for temperate climates (15-25°C) and normal pH (≤7); this is based on the ideal temperature for tomatoes being within 17-21°C and the ideal nutrient solution pH as 5.5-6.0 (Snyder, 2010; Langenhoven, 2018; Bayer, 2019). Note that these emission factors are also applied to all Georgia farms.

Using these categories, Table 52 provides the NH_3 emission factors used for mineral (synthetic) fertiliser applications ($EF_{NH3,M}$), taken directly from EEA 2019 Tier 2 guidance (EMEP & EEA, 2019a). These emission factors are assumed to have an uncertainty range of +/- 50%.

***Table 52** – NH_3 emission factors for mineral fertiliser application, in kg NH_3 (kg N applied)$^{-1}$*

Fertiliser type	**$EF_{NH3,M}$ for UK soil-based farms** [cool climate, normal pH]	**$EF_{NH3,M}$ for Georgia farms & UK soilless crops** [temperate climate, normal pH]
Anhydrous ammonia	0.019	0.020
Ammonium nitrate	0.015	0.016
Ammonium phosphate	0.050	0.051
Ammonium sulphate	0.090	0.092
Calcium ammonium nitrate	0.008	0.008
NK mixtures	0.015	0.022
NPK mixtures	0.050	0.067
NP mixtures	0.050	0.067
N solutions	0.098	0.100
Calcium nitrate, and other straight N compounds	0.010	0.014
Urea	0.155	0.159

These emission factors are then applied to calculate NH_3 emissions from synthetic fertiliser application using the simple calculation shown in Equation 8.

$$E_{NH3,M} = \sum_{i=1}^{I}[(F_{SN})_i * (EF_{NH3,M})_{i,j}]$$

Equation 8 – NH_3 emissions to air from N volatilisation of synthetic N fertilisers

Where,

- $E_{NH3,M}$ = the emission of NH_3 from N in all mineral (synthetic) fertilisers applied for a specific crop, in [kg NH_3].
- $(F_{SN})_i$ = the amount of N in the specific synthetic fertiliser *i* applied to soil, out of the total number of fertilisers (*I*) in [kg N].

- $(EF_{NH3,M})_{i,j}$ = the emission factor for the emission of NH_3 from a specific mineral fertiliser *i*, for a specific geography, *j* (Georgia, USA, or UK), in [kg NH_3 (kg N input)$^{-1}$], as provided in Table 52 (EMEP & EEA, 2019a).

2.1.9.3.1.1 Uncertainty for soilless crops

There is some uncertainty on whether or not NH_3 emissions should be included at all for soilless crops, as there is little evidence that these emissions occur. Indeed, in the recommended emission factors as provided by Pluimers *et al.* (2000) for glasshouse rockwool cultivation, no emission factor was provided for NH_3. In addition, immediate NH_3 emissions from fertiliser application only occur for ammonium-based fertilisers, and since hydroponic nutrient solutions are mostly nitrate-based (as assumed in this study), these NH_3 emissions are likely relatively low. However, the EEA emission factors will still be applied for specific fertiliser types as used in this study, but a sensitivity analysis will be conducted to see the impact of the inclusion of these emissions.

2.1.9.3.2 Organic fertilisers

EEA guidance does not provide Tier 2 emission factors for the application of organic fertilisers other than manure (EMEP & EEA, 2019a). In addition, the guidance for estimating application emissions from manure depends upon a mass balance of N losses within the Total Ammoniacal Nitrogen (TAN) fraction of the manure, as this is the fraction that contributes to NH_3 emissions, over the different phases of manure management (losses while in yard / shed, losses during storage, and losses upon application). For the vegetable farmers applying manure in this study, little is known about the management of manure before it reaches the vegetable farm. Thus, this makes it difficult to use the emission factors provided in EEA 2019 guidance (EMEP & EEA, 2019b).

However, the AGRIBALYSE v.3.0 database provided an alternative method to estimate the application emissions from organic fertilisers (Koch and Thibault, 2020). Following the EEA 2019 guidance that NH_3 emissions occur solely from the TAN fraction of the manure (EMEP & EEA, 2019b), the AGRIBALYSE database also assumed this for other organic fertilisers (such as green waste composts).

The AGRIBALYSE methodology provides specific information for the TAN content of manures and organic fertilisers at the point of application; this thus takes into account any N losses that may have already occurred for manures, which is necessary to utilise EEA manure emission factors (Koch and Thibault, 2020). After defining the % TAN in each organic fertiliser at the point of application, AGRIBALYSE v.3.0 methodology then applies the emission factors provided by EEA 2016 guidance for the application of specific types of manures. The database uses the average emission factor for all manures as an estimate of the emission factor for other organic fertilisers. This same methodology is used within this LCA, applying the updated EEA 2019 guidance values (EMEP & EEA, 2019b).

TAN values for bought-in composts and organic fertilisers were taken from supplier-specific information, where possible. For manures, homemade composts, and other organic fertilisers that did not have supplier-specific information available, values are based on AGRIBALYSE v.3.0 guidance is used (Avadí *et al.*, 2020; Koch and Thibault, 2020). Using the data provided by AGRIBALYSE, the % of TAN per % total N for each organic fertiliser of interest was calculated, and then these factors were applied to the actual %N for specific fertilisers used by farmers in this study, to provide an estimate of %TAN in each fertiliser.

The general values of TAN (as a % out of total N), as calculated from AGRIBALYSE data (Avadí *et al.*, 2020; Koch and Thibault, 2020), and the emission factors used for the relevant organic fertilisers from EEA 2019 guidance (EMEP & EEA, 2019b), are provided in Table 53. The value given for average organic fertilisers is applied to non-manure fertilisers (including composts), only when supplier-specific information on ammoniacal concentration was not available.

Table 53 *– TAN (%) and NH_3 emission factors ($EF_{NH3,o}$) for organic fertiliser application*

Fertiliser type	**%TAN out of total N in fertiliser**	**Emission factor, $EF_{NH3,O}$ [kg NH_3-N (kg TAN)$^{-1}$]**
Cow manure	20	0.68
Horse manure	10.1	0.90
Free-range chicken manure	31.5	0.45
Rabbit manure	6.58	0.61[a]
Composted manure	5.0	0.61[a]
Green waste compost	10.4	0.61[a]
Average organic fertiliser	9.05	0.61[a]

[a] Calculated as the average emission factor for all listed solid animal manures (14 types), as per EMEP & EEA (2019b).

Using these emission factors and TAN values, NH_3 emissions for organic fertilisers and manures can be calculated using Equation 9.

$$E_{NH3,O} = \sum_{i=1}^{I}[(F_{ON})_i * \%TAN_i * (EF_{NH3,O})_i * (\frac{17}{14})]$$

Equation 9 - NH_3 emissions to air from N volatilisation of organic N fertilisers

Where,

- $E_{NH3,O}$ = the emission of NH_3 from N in all organic fertilisers applied for a specific crop, in [kg NH_3].
- $(F_{ON})_i$ = the amount of N in the specific organic fertiliser *i* applied to soil, out of the total number of fertilisers (*I*) in [kg N]; N contents of organic fertilisers are provided in Table 48.
- *%TAN* = the percent of Total Ammoniacal Nitrogen (TAN) per total N in a specific organic fertiliser *i*, as provided in Table 53.
- $(EF_{NH3,O})_i$ = the emission factor for the emission of NH_3 from a specific organic fertiliser *i*, in [kg NH_3-N (kg N input)$^{-1}$], as provided in Table 53.
- $\left(\frac{17}{14}\right)$ = conversion factor to convert [kg NH_3-N] to [kg NH_3]

2.1.9.4 Nitrogen oxide (NO_X) emission calculations

Nitrogen oxide (NO_x) emissions also occur during from volatilisation and resulting deposition of these gases on soil or water bodies, as for NH_3 (IPCC, 2019). For soils in which the pH is maintained above 5.0, as is generally the case for the soils considered in this study, NO_x emissions mainly come from the nitrification process, where ammonium (NH_4^+) is oxidised to nitrate (NO_3^-) by microorganisms (EMEP & EEA, 2019a). NO is produced as an intermediate product during this process. Increased nitrification is expected to occur after the application of ammonium-based fertilisers, incorporation of crop residues, or soil tillage (EMEP & EEA, 2019a). NO_x production from agricultural soils are influenced by mineral N concentration, temperature, soil C, and soil moisture.

Nitrogen oxide (NO_x) emissions are calculated based on the Tier 1 emission factor (designated as EF_{NO2} here) provided in EEA 2019 guidance, which is specified for synthetic fertilisers, manures and other organic waste material (EMEP & EEA, 2019a). These are applied to all seedling production and crop cultivation processes (both soil-based and soilless). No specific emission factors were applied for soilless cultivation due to a lack of available data on measured NO_x emissions from hydroponic systems. This is also the methodology used for all crop processes within ecoinvent v.3.0, WFLDB v.3.0, and AGRIBALYSE v.3.0 databases, although WFLDB and ecoinvent v.3.0 use older EEA 2013 values and AGRIBALYSE uses EEA 2009 guidelines values (Nemecek *et al.*, 2019; Koch and Thibault, 2020).

No Tier 2 emission factors for NO_x were available in the EEA 2019 guidance; thus, there is a higher level of uncertainty using the general Tier 1 emission factor. However, WFLDB methodology specified that the importance of NO_x emissions from the application of N fertilisers is relatively small compared to other global sources of NO_x emissions; thus, simple emission factors can be used (Nemecek *et al.*, 2019). Separate uncertainty ranges are applied for seedling production and soil-based cultivation vs. soilless seedling production and cultivation. The uncertainty range applied for soil-based cultivation is that as provided in EEA guidance, while that used for soilless crops is based on literature specific to soilless cultivation.

NO_x emissions are modelled as NO_2 emissions in this LCA study, concurrent with the emission factor provided in EEA guidance and with how NO_x emissions are modelled within ecoinvent v.3.0 and WFLDB v.3.0 crop processes. It should be noted that alternatively, AGRIBALYSE v.3.0 crop processes model NO_x emissions collectively as “nitrogen oxides”, while Agri-footprint v.4.0 processes use general IPCC emission factors which assume all volatilised N as NH_3 (and thus no NO_x emissions are modelled) (Durlinger *et al.*, 2017; Koch and Thibault, 2020).

Finally, NO_x emissions are calculated from total fertiliser N input using the provided EEA 2019 emission factor (see: Table 47), after subtracting the amount of NH_3-N volatilised from the total input N. This subtraction is not specified in EEA guidance; however, it is performed in both WFLDB v.3.0 and AGRIBALYSE v.3.0 methodology, and thus is applied here (Nemecek *et al.*, 2019; Koch and Thibault, 2020).

$$E_{NO2} = \; [(F_{SN} + F_{ON}) - \left(E_{NH3,S} + E_{NH3,O}\right) * \; (\frac{14}{17})] * EF_{NO2}$$

Equation 10 – NO_2 emissions to air from N volatilisation of N fertiliser application

Where,

- E_{NO2} = the emission of NO_x, modelled as NO_2, from all applied N in synthetic and organic fertilisers, in [kg NO_2].

- F_{SN} = the total amount of N in all applied synthetic fertilisers for a specific crop, in [kg N].
- F_{ON} = the total amount of N in all applied organic fertilisers for a specific crop, in [kg N]; values for all organic fertilisers are provided in Table 48.
- $E_{NH3,S}$ = the emission of NH_3 from N in all synthetic fertilisers applied for a specific crop, in [kg NH_3].
- $E_{NH3,O}$ = the emission of NH_3 from N in all organic fertilisers applied for a specific crop, in [kg NH_3].
- $\left(\frac{14}{17}\right)$ = conversion factor to convert [kg NH_3] to [kg NH_3-N]
- EF_{NO2} = 0.04, a constant emission factor for the emission of NO_x (modelled as NO_2), in [kg NO_2 (kg N applied)$^{-1}$]. This emission factor is applied to the total N input from fertiliser application after subtracting losses from NH_3 volatilisation (Nemecek *et al.*, 2019; Koch and Thibault, 2020).

2.1.9.4.1 Uncertainty for soilless crops

There is great variability in the emission factors applied for NO_x emissions in soilless crop LCAs. As noted, WFLDB v.3.0, ecoinvent v.3.0, and AGRIBALYSE v.3.0 again use EEA Tier 1 emission factors (assuming 1.2% NO_x-N per kg N applied), after subtracting NH_3-N emissions. Little available evidence exists for measured NO_x emissions from soilless glasshouse cultivation. Hosono *et al.* (2006) measured an average 0.47% of NO-N emitted per total N for tomato glasshouse cultivation, although these were grown in soil. Alternatively, Montero *et al.* (2011) assumed 0.125% NO_x-N per kg applied N, based on guidance by Audsley *et al.* (1997) (not specific to soilless crops). Finally, Pluimers *et al.* (2000) provides an emission factor of 2.5% NO_x-N volatilised per applied N; while this appears higher than other factors, it should be noted that in contrast, they also did not assume any NH_3 emissions.

Based on these studies, an uncertainty range will be employed based on the other emission factors discussed here (0.125-2.5% NO_x-N, consequently 0.41-8.2% NO_2 per applied N), and an analysis will be conducted to test the sensitivity of results to NO_x emission levels.

2.1.9.5 Nitrate leaching and runoff calculations

N leaching and runoff from agricultural soils can result in emissions to groundwater and surface water (respectively), primarily with N in the nitrate (NO_3^-) form. NO_3^- is very mobile while other N forms, such as ammonium (NH_4^+), generally do not leach. Even fertiliser N that is not in the NO_3^- form can be converted to nitrate by microorganisms via mineralisation (conversion of organic N to nitrate-N) or nitrification (conversion of ammonium to nitrate), thus becoming available for leaching (Wyatt, Arnall and Ochsner, 2019).

Surface runoff can occur if rainfall exceeds the maximum water infiltration level in the soil (Velthof *et al.*, 2009); thus soil and any soluble salts will be carried with the water that runs off the surface. On the other hand, leaching occurs when the soil is unable to hold any more water (soil field capacity is surpassed), and gravity moves the water and any dissolved salts down the soil profile. If nitrate is present in the soil in excess of biological demand, this can leach through the soil profile into groundwater, depending on underlying soil or bedrock conditions.

Nitrate leaching and runoff can lead to both environmental and public health risks. When nitrate that is leached in groundwater comes up to the surface, and when nitrate runoff reaches surface water bodies, it contributes to eutrophication (Nemecek, Schnetzer and Reinhard, 2016). Nitrate leaching from agricultural fields is the largest anthropogenic N input into marine environments globally (Steffen *et al.*, 2015). Once leached, nitrate also induces N_2O emissions, thus contributing to climate change impacts (IPCC, 2019). Finally, nitrate in ground water can potentially be converted to nitrite; if this is used as drinking water, it can have an acute toxic and carcinogenic effect for humans (Ward *et al.*, 2018).

In this study, nitrate leaching is an important factor to consider because of its risk on the sites included in this study and because of its high influence on impact assessment categories (namely, the marine eutrophication and global warming). In Georgia, all but the south-eastern portion of the state exhibits some risk of N leaching due to agricultural N additions, a high percentage of well-drained soils, and a higher minimum thickness of the unsaturated soil zone (lower potential for denitrification) (Nolan, Hitt and Ruddy, 2002). In addition, the hot and humid temperatures in the state can cause the soil to remain wet or moist, thus being more prone to leaching (Liu *et al.*, 2010). In the UK, 9 out of the 14 farms in the study were found to be within a nitrate vulnerable zone, defined as an area at risk from agricultural nitrate pollution (UK Environment Agency, 2021). In addition, a recent global meta-analysis found that nitrate leaching rates were highest for vegetable production (in contrast to other crop production) (Wang *et al.*, 2019). It was hypothesised that this was because of the low-efficiency irrigation, frequent cultivation of many annual crop cycles, and practices of over-fertilisation that often accompany vegetable cultivation (Wang *et al.*, 2019).

Nitrate leaching can be influenced by many site-dependent factors. High leaching risk occurs in areas with high precipitation rates, low evapotranspiration, high N inputs (often from agriculture), well-draining soils, and shallow rooting depths. Runoff is similarly influenced by precipitation, as well as infiltration rates of the soil and slope of the field. Precipitation and evapotranspiration are important factors as leaching and runoff occur through a movement of water. Higher potential evapotranspiration rates (i.e., the water being removed from the soil through evaporation and plant transpiration) reduce leaching as water is removed from the soil surface.

Additionally, nitrate leaching is very sensitive to N inputs. A recent global meta-analysis concluded that nitrate leaching rates in agricultural systems was exponentially related to increasing N fertiliser inputs per area (Wang *et al.*, 2019). The meta-analysis found that only small amounts of nitrate would leach if N fertiliser applications were less than that required for maximum crop yield, but that leaching increases rapidly once over-fertilisation occurs (Wang *et al.*, 2019). Type of fertiliser is also important, with highly soluble fertilisers, such as ammonium nitrate, often leading to higher levels of nitrate leaching during a shorter period of time.

Nitrate leaching is also influenced by other soil-related factors, such as texture, organic N and C content, and soil profile depth (maximum rooting depth). Higher leaching rates were reported on soils with higher soil organic carbon (SOC) and total nitrogen (TN), likely because a higher organic matter and nitrogen content accelerates mineralisation and thus increases the nitrate-N pool available for leaching (Wang *et al.*, 2019). Water-holding capacity of the soil, influenced by soil texture, also plays a role. Soils with higher clay contents often exhibit lower levels of leaching, as they can hold more water compared to sandy soils which drain more readily (Killpack and Buchholz, 1993).

Modelling nitrate leaching is an important challenge in LCAs. A review of N emission models, applied to oil palm plantations, found that leaching was the most important N emission pathway, accounting for as much as 80% of total N losses (Pardon *et al.*, 2016). The study also found that N leaching was one of the most variable emission pathways to model, when comparing leaching results from eight different sub-models (Pardon *et al.*, 2016). Another recent study compared seven different nitrate leaching models when modelling impacts from wheat cultivation; models were selected as those which were medium effort and perhaps the most accessible to an LCA practitioner (Henryson *et al.*, 2020). The study found that N leaching varied by five-fold and contributed to 47-93% of the total eutrophication potential, depending on the model used. Avadí *et al.* (2022) similarly found high discrepancy between nitrate leaching models for different crop processes in LCA databases. Many of these nitrate leaching models also show extreme sensitivity to certain inputs.

Thus, nitrate modelling is a very important factor to consider for agricultural emissions, but it is difficult to model accurately. A wide array of models exist. In some cases, an all-encompassing emission factor is applied based on literature review, such as in the IPCC guidelines (IPCC, 2019). The updated IPCC emission factor estimates 24% of N leached as NO_3-N per kg N applied (in wet climates), thus modelling nitrate leaching as a linear function of N input. The sensitivity of N leaching to over-fertilisation will be lost in these types of linear models.

Other site-dependent models may require data inputs related to soil properties and rainfall, applying these either within regression equations, such as the SQCB-NO_3 model and Smaling 1993 model (Smaling, Stoorvogel and Windmeijer, 1993; Emmenegger, Reinhard and Zah, 2009), or within corrected emission factors, such as in MITERRA, SALCA-NO_3, and the Poore-Nemecek emission factors (Velthof *et al.*, 2009; Poore and Nemecek, 2018b; Nemecek *et al.*, 2019). Other models may require even more detailed input, such as dates of fertiliser application, method of application, and water flow modules (e.g., considering potential evapotranspiration).

In Henryson *et al. (*2020)'s comparison of five nitrate leaching models, they found that all models that utilised site-dependent data estimated lower N leaching rates than the site-generic IPCC Tier 1 emission factor; thus, they recommended using any model requiring site-specific information as a preference. However, due to the variability between the different models and the limitations imposed by each model, they could not recommend one site-dependent model over any other for use. There is thus a trade-off between efficient modelling of N-related emissions; the availability and accessibility of data needed to fit these models; and the time it takes to use a specific model.

2.1.9.5.1 Nitrate emissions to water for soil-based crop cultivation

In this study, N leaching in soil-based systems may occur from the loss of N in the application of fertilisers or incorporation of crop residues, as well as from the mineralisation of N during the loss of soil carbon associated with land use change or management practices.

The latter (N mineralisation) is not individually considered in this LCA, as it is assumed to be very low for non-organic soils, such as the ones being considered here; further, no major land use changes are considered, consistent with Agri-footprint v.4.0 methodology (Durlinger *et al.*, 2017).

In this LCA, several models were screened for the calculation of NO_3^- emissions from soil-based systems (with soilless systems considered separately). Five models were ultimately selected for further exploration. This includes the site-generic IPCC 2019 Tier 1 emission factor (IPCC, 2019), as well as four other site-dependent models: the de Willigen 2000 model (de Willigen, 2000); the SQCB-NO_3 model (Emmenegger, Reinhard and Zah, 2009); the Smaling 1993 model (Smaling, Stoorvogel and Windmeijer, 1993); and the Poore-Nemecek model (Poore and Nemecek, 2018b). These five models were selected based on their use in large-scale LCA studies and LCI databases; their global geographical scope; and the accessibility of data needed to use the models. Other models such as SALCA-NO_3 (used in ecoinvent and WFLDB databases) and MITERRA-Europe were considered, but not applied because of their validity for only European contexts (Velthof *et al.*, 2009; Nemecek *et al.*, 2019).

It should be noted that all site-dependent models only estimate nitrate emissions from leaching, assuming emissions from runoff to be negligible at low slopes. Only the generic IPCC emission factor provides an estimate for both leaching and runoff. As an example, the MITERRA-Europe model (not used here) considers runoff to be 0 for slopes less than 9% (Velthof *et al.*, 2009). In this study, it will be assumed that runoff is negligible as most farms in the study do not have a significant slope.

The default model used in this LCA is the de Willigen 2000 model, as it takes into account the most detailed data and is the most applicable for this LCA. Additionally, the equation has been qualified through an extensive literature review (Roy *et al.*, 2003) and has been used in a wide variety of large-scale LCA studies (Lesschen *et al.*, 2007; Mekonnen, Lutter and Martinez, 2016), including an FAO-based study to estimate N-related emissions in sub-Saharan Africa (Roy *et al.*, 2003).

The other four models are explored within the sensitivity analysis to determine how the model choice affects final results, particularly for the marine eutrophication and global warming impact categories. It should be noted that the SQCB-NO_3 model uses almost identical inputs as the de Willigen 2000 model, as it is based on the same basic regression equation developed by de Willigen; however, the de Willigen equation was chosen due to its increased specificity when determining the N available for leaching and because of its validation for academic studies (Roy *et al.*, 2003). The SQCB-NO_3 model, whilst used in various LCA studies and major LCI databases (e.g., ecoinvent, WFLDB, AGRIBALYSE), is not advised for academic use (Emmenegger, Reinhard and Zah, 2009). The de Willigen model was also chosen over the Poore-Nemecek 2018 model; Smaling 1993 model; and the IPCC emission factor as, in contrast to the other models, it considers the net N flux in the soil, thus taking into account the sensitivity of N leaching to under- and over-fertilisation. However, none of the models are perfect, and each comes with its own set of challenges and limitations.

The de Willigen regression equation estimates nitrate emissions by calculating the amount of N available for leaching and then applying a factor that determines the magnitude of leaching. The amount of N available for leaching is based on the net N flux, calculated using inputs of applied fertiliser N and mineralised organic N in the soil (as a function of soil C content) and an output of N uptake in the crop. This flux is then multiplied by a factor that determines the fraction of mobile N that will be leached (Smaling *et al.*, 2008). This factor is directly related to the average annual precipitation and inversely related to clay content of the soil and crop rooting depth. This regression equation has been validated for the conditions of: annual rainfall between 40 mm and 2000 mm; clay content between 3% and 54%; and crop rooting depth between 0.25 m and 2 m (Emmenegger, Reinhard and Zah, 2009). The crop cultivation processes modelled within this LCA fall within the scope of this model.

The model requires specific site-specific data for the equation, including: applied fertiliser N; N uptake in the crop; annual precipitation rates; irrigation rates; clay content of the soil (%); and total soil organic nitrogen (kg ha^{-1}). Applied fertiliser N and irrigation rates are taken from farm lifecycle inventories and crop N uptake is estimated as the N in all exported crop and crop residues, using values provided in Table 51. Precipitation, clay content, and total soil organic nitrogen for the farm sites were estimated using published meteorological data and soil maps, using the most recent and local data available (see: Table 54 and Table 55).

The de Willigen equation is given as Equation 11. The original equation has been slightly updated to better suit the needs of this study. The equation provides an output of nitrate leaching per total crop cultivation area on each farm site (E_{NO3}), in kg NO_3^-, rather than per individual hectare, by using the total N flux for the gross area of crop cultivation and calculating the organic N mineralisation for this area. Further, water from irrigation (in mm) has been added to the annual average rainfall value, as done in the later adaptation of this model (SQCB-NO_3). This is an important addition as the majority of farms considered in this study use irrigation, which can contribute to leaching. The N in cover crop residues is also considered as an additional N fertility input when a specific cover crop is used to provide fertility to a subsequent crop in a rotation. However, main crop residues which are incorporated into the soil are not considered as an N input to avoid double counting, as they do not leave the system. Finally, an allocation factor (A_{ag}) has been applied (Mekonnen, Lutter and Martinez, 2016). This factor allocates the total nitrate emissions to only anthropogenic inputs on the farm, excluding the contribution from mineralised organic matter. This allocation is applied since mineralised N is not accounted for in other emission calculations, and thus it is not included here to maintain consistency throughout the LCA model. The calculation for the allocation factor is provided in Equation 12.

There are also some additional limits set for the de Willigen equation. The first part of the equation (in brackets) is set at a maximum value of 1 to prevent any overestimate of leaching (Smaling *et al.*, 2008); this is because it is assumed that leached N cannot exceed the amount of N available for leaching, as calculated from the N flux (the second part of the equation, in brackets). Also, if the equation produces a negative value (because crop N uptake is higher than the N inputs), then the leached emission value is set to 0, and it is assumed that no leaching occurs.

Finally, there are specific considerations for crop cultivation indoors (e.g., in polytunnels and glasshouses) and for certain irrigation types. Average annual precipitation ($P_{f,c}$) is set at 0 for any crops cultivated under cover. Further, the water from irrigation ($I_{f,c}$) is not considered for drip-irrigated crops, in line with IPCC guidance, which does not consider drip-irrigated crops to contribute to leaching risk (IPCC, 2019). Thus, for crops cultivated only indoors with drip irrigation, leaching is assumed to be 0. For crop lifecycles that include some crops cultivated indoors and some outdoors (on the same farm), leaching values will only be applied to the portion of crop cultivation that has leaching risk. For example, if some tomatoes are cultivated indoors in polytunnels with drip irrigation, and some outdoors, then the model will only be applied to the outdoor portion of crop cultivation (considering any relevant irrigation and N fluxes for that portion only).

$$E_{NO3-S} = \left[0.0463 + \left(0.0037 * \frac{P_{f,c}+I_{f,c}}{C_f * L_c}\right)\right] * \left[(F_{SN} + F_{ON} + F_{CC}) + \left(N_{OM} * a_f * DR\right) - U\right] * A_{ag} * \left(\frac{62}{14}\right)$$

Equation 11 – de Willigen 2000 model for NO_3^- emissions to water from N inputs to soil

$$A_{ag} = \frac{F_{SN} + F_{ON} + F_{CC}}{(F_{SN} + F_{ON} + F_{CC}) + (N_{OM} * a_f * DR)}$$

Equation 12 – Allocation factor used in the de Willigen (2000) nitrate leaching model

Where,

- $E_{NO3\text{-}S}$ = the total emission of nitrate during the crop cultivation for a specific farm, in [kg NO_3^- (total crop cultivation area in ha)$^{-1}$] - *not* as [kg NO_3^- (ha)$^{-1}$].
  P_f = the average annual precipitation that applies to the crop lifecycle *c*, grown on a specific farm site, *f*, in [mm yr^{-1}]. Values are provided in Table 54 for U.S. farms and Table 55 for UK farms. This value is assumed as 0 for crops cultivated under cover (e.g., in polytunnels or greenhouses).
- I_f = the applied amount of irrigation water during the cultivation phase for a specific crop *c*, grown on a specific farm site *f*, in [mm or L m^{-2}]. This value is set to 0 for any drip-irrigated crops, as drip irrigation is assumed not to significantly contribute to leaching, based on IPCC guidance (IPCC, 2019).
  C_f = the clay content of the soil at a specific farm site *f*, in [% clay] as specified in Table 54 for U.S. farms and Table 55 for UK farms. This value is included in the calculation in percent form (base 100), not as a fraction.
- L_c = the maximum rooting depth of a specific crop, *c*, in [m]. For tomatoes, the rooting depth is assumed as 1 m (Machado and Oliveira, 2005) and for kale as 1.8 m (Hassan, Dresbøll and Thorup-Kristensen, 2021).
- F_{SN} = the total applied nitrogen from all synthetic (mineral) fertilisers for a specific crop, in [kg N].
- F_{ON} = the total applied nitrogen from all organic fertilisers for a specific crop, in [kg N].
- F_{CC} = the amount of N in cover crop residues that are returned to soil as a fertility input for the crop lifecycle of interest, in [kg N], calculating using Equation 4.
- N_{OM} = the soil organic nitrogen content for a specific farm site, *f*, in [kg organic N (ha)$^{-1}$]. This is calculated using Equation 13 based on input parameters as described in Table 54 for U.S. farms and Table 55 for UK farms.

- a_f = the gross area of cultivation on a specific farm site, *f*, for a specific crop lifecycle, *c*, in [ha]. This is applied to the N_{OM} value (which is in kg organic N per ha) to deduce the total amount of organic N per a particular farm site, since the rest of the N flux is based on total inputs and outputs for the particular crop of interest.
- DR = 0.016; the decomposition rate of N in organic matter. This is based on an assumed value of 1.6%, as used in other applications of the model (Mekonnen, Lutter and Martinez, 2016; Nemecek *et al.*, 2019).
- U = the nitrogen uptake of the crop, *c*, in [kg N]. This includes the total N uptake from all crop biomass that is exported from the field, thus excluding any crop residues that are re-incorporated to the soil. This includes residues sent for animal feed or composting, as well as the harvested crop. Calculations of N in crops and crop residues are provided in Equation 5, Equation 6, and Equation 7.
- A_{ag} = the allocation factor attributing N leaching to anthropogenic agricultural N inputs, in [kg N in fertilisers and cover crop residues (kg N mineralised)$^{-1}$]. This factor allocates the total nitrate emissions to anthropogenic inputs on the farm, excluding the contribution from mineralised organic matter.
- $\left(\frac{62}{14}\right)$ = conversion factor to convert [kg NO_3-N] to [kg NO_3^-]

The input data required for the de Willigen 2000 model, and the calculation of TON as used in the model, is provided in Table 54 for U.S. farms and Table 55 for the UK farms. Annual average precipitation is based on the 2019 year (the study year); for the US, this data derived from county-level average annual precipitation rates (NOAA, 2022), and for the UK farms, it is based on regional data on average annual rainfall (UK Met Office, 2022a). Soil property data is considered for the 0-30cm depth, where possible, as done in the SQCB-NO_3 model, which uses an adapted form of the de Willigen equation (Emmenegger, Reinhard and Zah, 2009). Soil properties of SOC, C/N content, and bulk density are readily available in national datasets, and these are used to calculate the soil organic nitrogen (SON), which is not generally published in these datasets.

For the U.S. farms, soil property data is provided by the 2021 national soil surveys, which are performed on a county level and published in the Web Soil Survey (NRCS, 2021). Soil data can be found for each farm site of interest using the Web Soil Survey mapping tool, and soil properties are provided based on the dominant soil series type for each farm and specific inputs of soil depth (30 cm). From this data, values for soil clay content (%), total organic carbon (%), and bulk density (g cm^{-3}) have been found; note that C/N values were not available through national soil survey data, and thus a generic C/N value of 11 has been used to approximate for all U.S. farms, as done in WFLDB v.3.0 methodology (Nemecek *et al.*, 2019).

For the UK farms, soil properties are taken from a range of national study data, all available through the UK Soil Observatory map viewer (UK Soil Observatory, 2022b). Topsoil carbon content (0-30 cm depth) is provided from data by the National Soil Resources Institute (Bradley *et al.*, 2005), while bulk density and C/N ratio (only available for 0-15 cm depth) was provided through the 2007 Countryside Survey (UK Soil Observatory, 2007; Emmett *et al.*, 2010). Clay content (%) was not provided directly, but was estimated from soil texture designations defined in the British Geological Survey (British Geological Survey, 2021; UK Soil Observatory, 2022a), utilising the soil texture triangle as per Lawley (2009).

__Table 54__ – Input data for de Willigen (2000) leaching model, for U.S. farms

U.S. farm	Average annual precipitation [mm] [a]	Soil clay content [%] [b]	Soil total organic carbon content (TOC) [%] [b]	Soil bulk density [g cm^{-3}] [b]
U-O-1	1,276	14.0	0.58	1.38
PU-O-1	1,276	25.3	0.44	1.52
PU-O-2	1,238	25.8	0.44	1.53
PU-O-3	1,240	10.3	0.87	1.49
PU-O-4	1,372	18.8	0.24	1.45
R-O-1	1,189	10.8	0.49	1.54
R-O-2	1,090	38.0	0.34	1.35
R-O-3	1,379	30.0	0.44	1.57
R-C-1	997	7.40	0.42	1.62
R-C-2	1,958	29.8	0.45	1.33
R-C-3	1,074	7.40	0.45	1.62

[a] This is taken from county-level data for 2019, provided by (NOAA, 2022).
[b] Soil property data (clay content, total organic carbon content, and bulk density) is provided for each farm site using national soil survey data, provided in the NRCS Web Soil Survey (NRCS, 2021). The properties are provided for specific soil series, and the map viewer allows one to view the most abundant soil series type on each farm. The property data has been provided for a depth range of 0-30 cm.

Table 55 *– Input data for de Willigen (2000) leaching model, for UK farms*

UK farm	Average annual precipitation [mm] [a]	Soil clay content [%] [b]	Soil total organic carbon content (TOC) [%] [c]	Soil carbon to nitrogen ratio, C/N [d]	Soil bulk density [g cm^{-3}] [d]
U-O-1	1,391	5.0	7.98	10.61	1.18
U-O-2	1,391	5.0	7.98	10.61	1.18
PU-O-1	925	15	6.00	11.04	0.96
PU-O-2	925	15	5.41	11.04	1.18
PU-O-3	925	18	7.78	11.94	1.22
PU-O-4	925	18	7.36	11.51	1.16
R-O-1	1,502	18	5.07	11.94	1.16
R-O-2	859	18	6.17	10.61	1.18
R-C-1	973	18	7.36	11.23	1.16
R-C-2	925	5.0	4.76	11.23	1.16
R-C-3	1,502	11.5	7.81	11.23	1.16
R-C-4	1,391	18	5.91	10.61	1.18
R-C-5	628	18	6.76	11.51	1.16

[a] This is taken from regional-level data for 2019, provided by (UK Met Office, 2022a). Note that these are actual annual average rainfall values, but assume they are roughly equivalent to annual precipitation values (as snowfall is not extremely common in this area).
[b] Clay content (%) has been personally estimated, using the soil texture designations defined in the British Geological Survey (British Geological Survey, 2021; UK Soil Observatory, 2022a), and applying the soil texture triangle as per (Lawley, 2009).
[c] Topsoil carbon content (0-30 cm depth) is provided from data by the National Soil Resources Institute (Bradley *et al.*, 2005), viewed in the UKSO map viewer for each farm site (UK Soil Observatory, 2022b).
[d] Soil C/N and bulk density values are provided from the 2007 Countryside Survey (Emmett *et al*., 2010), viewed in the UKSO map viewer. Note that this data is only available for the 0-15 cm depth, and these values are assumed for the entire 0-30 cm depth considered in the nitrate leaching model.

Soil organic nitrogen (SON) is then calculated using the TOC (total organic carbon) content, C/N ratio (organic C to total N), and bulk density (kg m^{-3}), assuming a soil volume of 3000 m^3 (considering 0-30 cm depth over 1 ha) and a ratio of organic N to total N as 0.85, as per WFLDB v.3.0 guidance (Nemecek *et al.*, 2019). Equation 13 details this calculation.

$$N_{OM} = \frac{TOC}{C/N} * r_{Norg} * V * BD$$

Equation 13 – Total organic N for specific farm sites

Where,

- N_{OM} = the soil organic nitrogen content for a specific farm site, in [kg organic N $(ha)^{-1}$].
- TOC = the total soil organic carbon content for a specific farm site, in [kg organic C $(kg\ soil)^{-1}$].
- C/N = the carbon to nitrogen ratio for a specific farm site, in [kg organic C (kg total $N)^{-1}$].
- r_{Norg} = 0.85; this is the ratio of organic N to total N, in [kg organic N (kg total $N)^{-1}$]. A constant value of 0.85 is assumed, as per WFLDB guidance (Nemecek *et al.*, 2019).
- V = 3000; this is the soil volume considered, in [m^3 soil]. This constant value is based on a 30 cm soil depth over 1 ha (10,000 m^2). Different studies consider different soil depths, however, 30 cm was considered as this is the default depth used in the SCQB-NO_3 model (Emmenegger, Reinhard and Zah, 2009).
- BD = the soil bulk density at a specific farm site, in [kg (m^{-3})].

2.1.9.5.2 Nitrate leaching emission models explored in sensitivity analysis

Four other nitrate leaching models were explored within this LCA for soil-based cultivation. The closest model to the de Willigen 2000 model is the SQCB (Sustainability Quick Check for Biofuels) model (Emmenegger, Reinhard and Zah, 2009), as it uses an adapted equation from (de Willigen, 2000; Roy *et al.*, 2003). Despite the SQCB-NO_3 model not being advised for academic use (Emmenegger, Reinhard and Zah, 2009), it has been used to estimate nitrate emissions in WFLDB v.3.0 and ecoinvent 3.0 databases for non-European countries; in AGRIBALYSE v.3.0 databases for certain crops (such as orchards, grapevines and certain vegetables); and in the open-source Crop.LCA tool, which generates LCAs of cropping systems (Goglio *et al.*, 2018).

The equation used in the SQCB-NO_3 model is provided in Equation 14. Note that included variables (as well as the units used) are identical to those used in the de Willigen (2000) model, given in Equation 11. The equation has been updated slightly so that the output is the given as total nitrate emission (kg NO_3) for a specific crop lifecycle, over the gross cultivation area (in ha).

$$E_{NO3-S} = \left(21.37 + \frac{P_f + I_{f,c}}{C_f * L_c} * \left[0.0037 * \frac{(F_{SN} + F_{ON} + F_{CC})}{a_f} + 0.000061 * C_{org,f} - \frac{0.00362 * U}{a_f}\right]\right) * \left(\frac{62}{14}\right) * a_f$$

Equation 14 – SQCB model for NO_3^- emissions to water from N inputs to soil

However, this model has its challenges. For one, the original equations (de Willigen, 2000) have been adapted in various forms across different studies and in LCA databases, and thus there is inconsistency associated with their application. For example, certain versions of the equation use inputs of organic N (Nemecek *et al.*, 2019), while others use organic C content to estimate N mineralisation (Roy *et al.*, 2003; Emmenegger, Reinhard and Zah, 2009).

Additionally, the format of the equation raises some doubts. If the N flux (N inputs minus N outputs) generates a negative result, then one would assume that the nitrate emission output from the equation would be zero. However, based on the format of the equation, a negative N flux multiplied by a factor that represents high risk of leaching (e.g., high precipitation and low clay content) would result in a more negative output; on the other hand, a negative N flux multiplied by a low factor of leaching might actually generate a positive nitrate leaching emission value, due to the added 21.37 constant. This creates some uncertainty in how this adapted equation was derived and the assumptions set in place. It has been noted that the equation is calibrated for adequate fertilisation, and thus is very sensitive to over- and under-fertilisation, generating negative values in cases of under-fertilisation and extremely high values in cases of over-fertilisation (Avadí *et al.*, 2022). The model is also very sensitive to inputs of clay content, nitrogen uptake, and rooting depth (Pardon *et al.*, 2016).

The Smaling model also uses basic regression equations based on average annual rainfall, N inputs from fertilisers, and N from organic matter mineralisation. However, it does not explicitly calculate the N flux, as N uptake in the crop is not considered. Three sets of equations are offered based on the clay content of the soil (<35%, 35-55%, and >55%). These equations were recommended for use by Roy *et al.* (2003) for farm-level N calculations as a potential alternative to the de Willigen (2000) equations; however, later guidance on country-wide N emission modelling gave preference to the de Willigen 2000 equations (Lesschen *et al.*, 2007). The Smaling 1993 equations are also used by Brockmann, Pradel and Hélias (2018) to model nitrate leaching emissions from organic fertilisers, manure, and crop residues.

For the farms of interest in this study, the first equation (for clay contents <35%) is mainly used; the second equation, for clay contents between 35-55%, is only used for one farm. Adapted versions of the first two Smaling 1993 equations, as used by Brockmann, Pradel and Hélias (2018), are provided in Equation 15. Note that the factor for organic N mineralisation (N_{min}) is directly set to zero, so that only anthropogenic inputs are considered (Brockmann, Pradel and Hélias, 2018). This maintains consistency with other models. All other variables within the equation are in the same form, and applied in the same fashion, as in the de Willigen equation (Equation 11). Again, the output of the equation provides the total emission of nitrate per total crop cultivation area [kg NO_3^- (total ha of crop cultivation)$^{-1}$].

$$E_{NO3-S} = \left[(F_{SN} + F_{ON} + F_{CC} + N_{min}) * \left(0.021 * (P_f + I_{f,c}) - 3.9\right)\right] * \frac{1}{100} * \frac{62}{14} \text{ for } C_f < 35\%$$

$$E_{NO3-S} = \left[(F_{SN} + F_{ON} + F_{CC} + N_{min}) * \left(0.014 * (P_f + I_{f,c}) + 0.71\right)\right] * \frac{1}{100} * \frac{62}{14} \text{ for } 35\% < C_f < 55\%$$

Equation 15 – Smaling 1993 model for NO_3^- emissions to water from N inputs to soil

The model created by Poore & Nemecek in 2018 is based on an extensive literature review, including approximately 91 field-based studies on nitrate leaching (Poore and Nemecek, 2018b). From these studies, nitrate leaching emission factors were created for "low" and "high" risk conditions, as well as a general emission factor for all other conditions, or in cases where both high and low risk conditions occurred. The risk categories were based on maximum rooting depth of the crop; clay content; and average annual precipitation. The emission factors are then applied to the N input to approximate the amount of leached mineral nitrogen. This is shown in Equation 16. The same conditions are applied to this model as all others. Precipitation and irrigation are considered to be 0 for indoor and drip-irrigated cultivation, respectively. The same N inputs as for the aforementioned models are considered, including synthetic fertilisers, organic fertilisers, and cover crop residues (F_{SN}, F_{ON}, and F_{CR}, respectively), in [kg N].

$$E_{NO3-S} = (F_{SN} + F_{ON} + F_{CR}) * \ Frac_{leach} * \left(\frac{62}{14}\right)$$

Equation 16 – Poore-Nemecek 2018 model for NO_3^- emissions to water from N inputs to soil

Where,

- E_{NO3-S} = the total emission of nitrate during the crop cultivation for a specific farm, in [kg NO_3^- (total farm area in ha)$^{-1}$]
- $Frac_{leach}$ = the emission factor developed for nitrate leaching, in [kg NO_3-N (kg N applied)$^{-1}$], based on the following conditions:
    - = 0.067, for a low risk of leaching, if at least one of the following conditions is true: maximum rooting depth > 1.3 m (this is true for all kale farms); soil clay content > 50%; average annual precipitation < 500mm (this is true for indoor growing, depending on the irrigation rate).
    - = 0.23, for a high risk of leaching, if at least one of the following conditions is true: maximum rooting depth < 0.5 m, soil sand content > 85%, or average annual precipitation > 1300 mm.
    - = 0.12, for all other conditions, or if both a high and low risk condition are met simultaneously.

Finally, the widely-used, generic IPCC Tier 1 emission factor for nitrate leaching was also modelled. This study uses the 2019 updated values to estimate the fraction of leached / runoff N from total N inputs (IPCC, 2019). As this is only one emission factor used across all farms, this is the least specific model employed. However, it does account for both leaching and runoff, whereas the prior four models only estimate leaching emissions. The IPCC emission factor assumes that 24% of the total N input is leached as NO_3-N (lowered slightly from the 2006 emission factor of 30%). However, the updated 2019 guidance specifies that for Tier 2 (country-specific) application of the emission factor, leaching should only be considered for regions where the average annual rainfall minus the reference evapotranspiration is greater than the soil water holding capacity, or where irrigation (except drip irrigation) is used. For all farms with outdoor field production in this study, the prior scenario is true and thus leaching is estimated for all outdoor field production.

Utilising these conditions, the IPCC 2019 generic emission factor is applied to the N inputs using Equation 17.

$$E_{NO3-S} = (F_{SN} + F_{ON} + F_{CR}) * \ Frac_{leach} * \left(\frac{62}{14}\right)$$

Equation 17 – IPCC 2019 model for NO_3^- emissions to water from N inputs to soil

Where,

- $E_{NO3\text{-}S}$ = the total emission of nitrate during the crop cultivation for a specific farm, in [kg NO_3^- (total farm area in ha)$^{-1}$]
- $Frac_{leach}$ = the emission factor developed for nitrate leaching, in [kg NO_3-N (kg N applied)$^{-1}$], based on the following conditions:
    - = 0.24 (uncertainty range: 0.01-0.73), for *wet* climates; where average annual rainfall minus the reference evapotranspiration is greater than the soil water holding capacity; or where irrigation (except drip irrigation) is used.
    - = 0, for dry climates, or all other cases that do not meet the above criteria.

2.1.9.5.2.1 Model uncertainties

Each model chosen comes with a set of limitations and challenges. Overall, one of the main issues with all of these models is their lack of consideration about the time (e.g., specific month) and method of fertiliser application, both of which can greatly impact leaching and runoff. The SALCA-NO_3 model, used by ecoinvent v.3.0 and WFLDB v.3.0 in European contexts, does take these into account, considering correction factors based on the season and type of fertiliser application (Richner *et al.*, 2014). However, this model could not be applied in this study as it is generated for use only in European contexts (originally modelled for Switzerland). Although all site-dependent models investigated in this study do consider the average annual precipitation, this does not provide an entirely accurate estimate of leaching, as rainfall varies greatly month-to-month in both U.S. and UK contexts. Potential evapotranspiration, which also greatly affects leaching rates, is also not taken into account in any of the models used; however, this is included in more complex models, such as INDIGO-N and STICS, which could not be employed here due to lack of accessible data (Bockstaller and Girardin, 2010; Coucheney *et al.*, 2015). The MITERRA-Europe model also considers potential evapotranspiration and could have been applied in this study, although it has only been verified for use in European contexts (Velthof *et al.*, 2009). However, Henryson *et al.*, (2020)'s comparison of N leaching models, which included both MITERRA and SQCB, found comparable results from the two, and all site-dependent models they assessed (which also included Poore-Nemecek) were equally recommended.

Another issue comes from how to regard N inputs in each model. The IPCC 2019 model specifies that the Tier 1 factor be applied to all N inputs, including synthetic and organic fertiliser N; N from crop residues; and N from soil mineralisation (which is not included for IPCC calculations in this study as there is assumed to be little change in soil organic matter from one year to the next). However, there is less specificity in how N inputs should be applied in the SQCB and Smaling equations. This has resulted in various adaptations being used, especially for the SQCB model.

For example, the original equation on which the SQCB model is based, as given by Roy *et al.* (2003), considers only mineral fertiliser N as the N input. However, the SQCB adaptation considers both mineral N and total N in organic N inputs. WFLDB v.3.0 guidance further specifies N inputs as mineral N in synthetic fertilisers and only the soluble N portion of organic fertilisers (Nemecek *et al.*, 2019). Finally, AGRIBALYSE v.3.0 considers mineral fertiliser N, organic fertiliser total N, and N input in crop residues (Koch and Thibault, 2020). As the SQCB model is especially sensitive to over-fertilisation and thus modelling of N inputs, this creates a challenge in how to apply this model. Further, it complicates how N uptake should be modelled. The SQCB model is supposedly estimating N uptake for the whole plant, but does not specify accounting for crop residues returned to the soil. In the Smaling equations, N uptake is not considered, and during use by Brockmann, Pradel and Hélias (2018), N inputs are seen to include total N from organic fertilisers.

It therefore unclear whether or not one should consider only the soluble fraction of N in organic inputs, or total N. On one hand, leaching can only occur from soluble forms of N, and thus this implies that only soluble forms of N should be considered, as done in WFLDB v.3.0 guidance (Nemecek *et al.*, 2019). However, organic N may still be mineralised into nitrate, thus becoming available for leaching. Usually, this occurs slowly over time, and subsequent crops may be able to take up the organic N that is mineralised each year. However, if excessive organic N inputs are applied each year, then organic N contents applied in prior years may be mineralised and result in excessive N available during the crop lifecycle of interest. In addition, if N mineralisation occurs after the crop lifecycle, before another crop is being grown on that land, then this could lead to even more leaching as there are no plants to take up the mineralised N. Thus, an additional sensitivity analysis has been conducted using the default leaching model (de Willigen) and assuming only soluble N from organic fertilisers, to see how this influences final impact assessment results.

2.1.9.5.3 Nitrate emissions to water for soilless crop cultivation

In soilless cultivation, nitrate emissions to water occur from the leachate (drained nutrient solution). Thus, leaching is considered only for open (draining) hydroponic systems. This applies to 50% of the hydroponic systems used for UK conventional tomato cultivation.

A figure of 45% NO_3-N per applied N was used as the emission factor for nitrate leached from open hydroponic systems ($EF_{NO3\text{-}SL}$). This was calculated as an average from studies that measured nitrate contents in leachate from tomato cultivation in open hydroponic systems. Values of NO_3-N per applied N were reported in relevant studies as: 48% for tomatoes grown in perlite (Sanjuan-Delmás *et al.*, 2020), and 29% (Thompson *et al.*, 2013) and 59% for tomatoes grown in rockwool (Massa *et al.*, 2010). An uncertainty range is employed based on the minimum and maximum leaching values measured in the studies used to derive the emission factor, which is 18-59% (Massa *et al.*, 2010; Thompson *et al.*, 2013; Sanjuan-Delmás *et al.*, 2020).

It should be noted that the emission factor calculated for use in this study (45% NO_3-N per applied N) is actually the same as that employed by Torrellas, Antón, López, *et al.* (2012) and by the WFLDB v.3.0 process for soilless tomato cultivation, based on guidance from Audsley *et al.* (1997).

The nitrate leaching emission factor for soilless crop production is applied to total N input in the nutrient solution, as per Equation 18.

$$E_{NO3-SL} = F_{SN} * EF_{NO3-SL} * (\frac{62}{14})$$

Equation 18 – Nitrate emissions to water for soilless crop cultivation (open hydroponic systems)

Where,

- E_{NO3-SL} = the emission of NO_3 to water for nitrate in nutrient solution leachate from open hydroponic systems (for soilless cultivation only), in [kg NO_3^-].
- F_{SN} = the total amount of N in all applied synthetic fertilisers for a specific crop, in [kg N].
- E_{NO3-SL} = 0.45, the emission factor of leached nitrate N from hydroponic systems, in [kg NO_3-N (kg applied N)$^{-1}$]. An uncertainty range of 0.18-0.59 is applied.
- $\left(\frac{62}{14}\right)$ = conversion factor to convert [kg NO_3-N] to [kg NO_3^-].

2.1.9.6 Nitrous oxide (N_2O) emission calculations

IPCC guidelines specify nitrous oxide emissions as occurring through both direct and indirect pathways. 'Direct' N_2O emissions refer to those occurring directly from the soils to which N is applied. Consequently, 'indirect' N_2O emissions refer to those occurring 'off-site', as a result of N volatilisation and then deposition from volatilised gases, as well as from N leaching (IPCC, 2019).

N_2O emissions are driven by the microbial N transformation processes of nitrification and denitrification. These transformations can occur for fertiliser N that is not immediately taken up by the plant. Nitrification describes the process of microbial oxidation of ammonium to nitrate in aerobic conditions, whereas denitrification is the microbial reduction of nitrate to nitrogen gas (N_2) under anaerobic conditions. N_2O is produced as a by-product of nitrification, leaking from microbial cells into the soil, and as an intermediate step in denitrification. These reactions are influenced by the availability of inorganic N in the soil, which is therefore affected by fertiliser N additions.

N_2O emissions particularly influence the global warming impact category, as well as stratospheric ozone depletion (although this is not reported on in this study). N_2O is a very potent greenhouse gas, with a global warming potential 298 times CO_2 on a 100-year scale as considered within the lifecycle impact assessment method (Huijbregts *et al.*, 2017), based on the IPCC Fourth Assessment report.

N_2O emissions in this LCA are modelled based on IPCC guidelines (IPCC, 2006c, 2019), which are also used in the LCA databases relevant to this study, such as ecoinvent v.3.0, WFLDB v.3.0, AGRIBALYSE v.3.0, and Agri-footprint v.4.0 (Durlinger *et al.*, 2017; Nemecek *et al.*, 2019; Koch and Thibault, 2020). The exception is for soilless cultivation and soilless seedling production, where a different emission factor based on measured data in literature is used for direct N_2O emissions. Soilless cultivation and soilless seedling production still use IPCC emission factors for indirect N_2O emissions.

The equations used to calculate direct and indirect emissions as used in this LCA and in the aforementioned LCI databases were defined in IPCC 2006 guidelines and carried through in the 2019 refinement (IPCC, 2006c, 2019). However, the 2019 refinement provided updated emission factors for N_2O emissions. The LCI databases previously mentioned (ecoinvent, WFLDB, AGRIBALYSE, and Agri-footprint) use the Tier 1 emission factors provided in the IPCC 2006 Guidelines (IPCC, 2006c). In this LCA, the updated emission factors provided in the IPCC 2019 refinement of guidelines are used instead (IPCC, 2019). Although this means discrepancy from factors employed by major databases, these emission factors were selected as they incorporate updated literature assessments, and further, the IPCC refinement has now provided disaggregated values for some Tier 1 emission factors based on climatic region (wet or dry) and type of fertiliser (synthetic or organic). Thus, emission factors can be selected that are more applicable to a particular farm and their choices of fertiliser inputs, which is important following the aim of this LCA being to compare environmental impacts based on farm type and management decisions.

Therefore, disaggregated IPCC Tier 1 emission factors are used in this LCA where possible. Emission factors for synthetic inputs are used for all mineral fertilisers, as well as mixes of mineral and organic fertilisers, whilst emission factors for organic inputs are used for organic fertilisers, manure, and crop residues, as per IPCC guidance (IPCC, 2019). Further, emission factors for wet climates are used for both UK and Georgia farms. Wet climates are defined by the IPCC as temperate and boreal zones where the ratio of annual precipitation: potential evapotranspiration is greater than 1. Both Georgia and the UK fall within this definition of wet climates when averaged over the year (IPCC, 2006b; Suleiman and Hoogenboom, 2007).

Calculations of N_2O emissions using IPCC equations include possible emissions from the following N sources: synthetic fertilisers; organic fertilisers (e.g., manure and compost); crop residues (from any cover crops and the main crop); urine and dung deposited on the soil by grazing animals; N mineralisation associated with the loss of soil organic matter, resulting from land use change or management of mineral soils; and the drainage of organic soils (i.e., histosols). However, the last three N sources are not accounted for in this LCA, because it is assumed that the cropland is not grazed (as confirmed by farmers in this study); that crops are cultivated on agricultural land, which will remain agricultural land for the foreseeable future, and any emissions from land use change are already counted for separately within the applied ecoinvent land transformation processes; and that the organic matter content of the soils will not substantially change year to year. These assumptions are concurrent with those applied in Agri-footprint v.4.0 crop processes (Durlinger *et al.*, 2017). Thus, the N contributions from these items will be excluded from the IPCC equations (IPCC, 2006c). The adjusted equations used in this study are included in this section.

Because this study uses farmer-specific data for fertiliser inputs, the main uncertainty in emissions calculations comes from the emission factors used and the calculations of leaching and volatilisation fractions. IPCC emission factors include uncertainty ranges based upon differences in local environmental conditions (e.g., climate, drainage ability, soil type, etc.).

2.1.9.6.1 Direct N_2O emissions for soil-based cultivation and seedling production

Direct N_2O emissions from soil result from an increase in available N, which enhances nitrification and denitrification rates, thereby increasing the production of N_2O (IPCC, 2019).

Direct N_2O emissions to air are calculated in Equation 19, based on N inputs from synthetic fertilisers, organic fertilisers, and crop residues. This equation is adapted version of that provided in IPCC guidelines (IPCC, 2006c, 2019). It uses two disaggregated emission factors for the application of synthetic and organic fertilisers in wet climates. The ability to use emission factors specified for synthetic and organic inputs allows for a more distinct representation of emissions from the different farm types and management practices used in this study. Indeed, the new IPCC emission factors direct N_2O emissions for organic fertilisers have been validated at least for manure application in the UK and have been confirmed as a better prediction for organic fertiliser emissions in contrast to the singular emission factor applied in IPCC 2006 guidelines (Thorman *et al.*, 2020)

$$E_{N2O,D-S} = [(F_{SN} * EF_{1,S}) + ((F_{ON} + F_{CR}) * EF_{1,O})] * \left(\frac{44}{28}\right)$$

Equation 19 – Direct N_2O emissions from N inputs for soil-based seedling and crop production

Where,

- $E_{N2O,D-S}$ = the direct emission of N_2O, from all applied N in synthetic and organic fertilisers and crop residues during crop cultivation and seedling production, in [kg N_2O].
- F_{SN} = the amount of N in synthetic fertilisers applied for a specific crop, in [kg N].
- F_{ON} = the amount of N in animal manure, compost, compost teas and other organic N additions applied to soils for a specific crop, in [kg N].
- F_{CR} = the amount of N in crop residues (both above-ground and below-ground) returned to soil, in [kg N], including both from the crop of interest as well as any cover crops or green manures used to improve the soil for that crop. Details of how this is calculated for cover crops and main crop residues is included previously in Section 2.1.9.2.2.
- $EF_{1,M}$ = 0.016, a constant emission factor for N_2O emissions from mineral N inputs in wet climates, in kg N_2O-N (kg N input)$^{-1}$ (IPCC, 2019). The uncertainty range of this value is 0.013-0.019, as it depends on climate, soil conditions, and agricultural management practices.
- $EF_{1,O}$ = 0.006, a constant emission factor for N_2O emissions from organic N inputs and crop residue N inputs, specified for wet climates, in kg N_2O-N (kg N input)$^{-1}$ (IPCC, 2019). The uncertainty range of this value is 0.001-0.011, as it depends on climate, soil conditions, and agricultural management practices.
- $\left(\frac{44}{28}\right)$ = conversion factor to convert [kg N_2O-N] to [kg N_2O].

2.1.9.6.2 Direct N_2O emissions for soilless cultivation and seedling production

In soil-based cultivation, N_2O emissions occur from both the processes of nitrification (conversion of ammonium to nitrate) and denitrification (conversion of nitrate to nitrogen gas). In soilless cultivation, emissions of N_2O are instead dominated by denitrification (Daum and Schenk, 1996b). This is likely due to the use of mainly nitrate-N in hydroponic nutrient solutions, as well as the fact that the inert conditions of soilless substrates and their low water retention capacity can inhibit nitrifying microbe population growth (Llorach-Massana *et al.*, 2017; Karlowsky *et al.*, 2021). Considering the fact that N-related emissions from soilless cultivation are dominated by different pathways than in soil-based cultivation, the decision was made to use a specific N_2O emission factor for soilless cultivation and seedling production.

After reviewing four different studies that directly measured N_2O emissions from soilless cultivation of cucumbers, tomatoes, and lettuce in rockwool, perlite, and coco coir substrates, an average emission factor of 0.87% N_2O-N per applied N was used (EF_{1-SL}). The emission factor was derived as an average of the following listed average emission factors (in %N_2O-N per applied N) reported by studies directly measuring N_2O emissions in soilless cultivation, including: (Daum and Schenk, 1996a) measuring 1.2% for cucumbers grown in rockwool; (Llorach-Massana *et al.*, 2017) measuring an average 0.5% for lettuce grown in perlite; (Yoshihara *et al.*, 2014) measuring an average 2.8% for tomatoes grown in rockwool; and (Karlowsky *et al.*, 2021) measuring 0.15% for cucumbers grown in rockwool, 0.31% for tomatoes grown in rockwool, and 0.46% for tomatoes grown in coco coir or perlite.

Using the minimum and maximum values provided by all these studies, an uncertainty range of 0.1-4.6% is used for the emission factor. This high range of uncertainty is likely because N_2O emissions in soilless cultivation depend on factors such as pH, N concentration in the nutrient solution, and even growing media (Daum and Schenk, 1996b, 1998; Karlowsky *et al.*, 2021). Acidic environments, higher N concentrations, and waterlogging of growing media can lead to higher N_2O emissions.

The derived emission factor is applied to the input N in nutrient solution, in a similar fashion to the calculation of direct N emissions for soil-based crops. It should be noted, however, that consideration of N input from crop residues is not included for soilless crops. This is because crop residues are not incorporated into the soil like in soil-based cultivation; rather, for the specific soilless lifecycle of interest (UK conventional tomato production), crop residues are composted. Thus, any emissions from crop residues are considered in composting processes in the respective scenarios and are not considered here.

Direct N_2O emissions for soilless cultivation and soilless seedling production is thus calculated simply using Equation 20.

$$E_{N2O,D-SL} = F_{SN} * \ EF_{1-SL} * \left(\frac{44}{28}\right)$$

Equation 20 - Direct N_2O emissions from N inputs, for soilless seedling and crop production

Where,

- $E_{N2O,D-SL}$ = the direct emission of N_2O, from all applied N in hydroponic nutrient solution during soilless crop cultivation and seedling production, in [kg N_2O].
- F_{SN} = the amount of N in synthetic fertilisers applied for soilless crop cultivation, in [kg N].
- EF_{1-SL} = 0.0087 (uncertainty range: 0.001-0.046), a constant emission factor for direct N_2O emissions from soilless cultivation, in [kg N_2O-N (kg N input)$^{-1}$].
- $\left(\frac{44}{28}\right)$ = conversion factor to convert [kg N_2O-N] to [kg N_2O].

2.1.9.6.2.1 Uncertainty for soilless crops

In contrast to defining a specific emission factor for soilless production, some LCAs and LCA databases instead simply employ emission factors as used for soil-based cultivation. Mosier *et al.* (1998) concluded that N_2O emissions from glasshouse crops were reasonably similar to soil-grown crops per unit of N input, and thus recommended using soil-based emission factors for N_2O. Thus, in many cases, the IPCC 2006/2019 generic Tier 1 emission factor for direct N_2O emissions from fertiliser application is used (1% N_2O-N per kg applied N). Indeed, this factor was decided to be used in AGRIBALYSE v.3.0 soilless crop processes after analysing a prior study measuring N_2O emissions from soilless cucumber production (Daum and Schenk, 1996a) and finding that emission factors from the study had great uncertainty but were generally similar to the IPCC 2006 emission factor. Similarly, this emission factor is employed in WFLDB v.3.0, ecoinvent v.3.0, AGRIBALYSE v.3.0, and other soilless LCA studies, such as (Röös and Karlsson, 2013).

It should be noted that the calculated emission factor used here (0.87% N_2O-N per applied N) is still similar to the 1% generic IPCC Tier 1 emission factor, the same of which was recommended by Pluimers *et al.* (2000) for soilless cultivation. However, it is nearly half of the disaggregated IPCC Tier 2 emission factor for synthetic fertiliser application in wet climates that is applied to all mineral fertilisers in this study (1.6% N_2O-N per applied N). It should be noted that even the 0.87% emission factor may be an overcalculation for soilless tomato cultivation in this study, as the only studies measuring N_2O emissions at scale, which include Llorach-Massana *et al.* (2017) in a rooftop greenhouse in Barcelona and Karlowsky *et al.* (2021) at a commercial glasshouse site in Germany, found much lower average emission factors (0.15-0.5%). Some LCA studies do not include N_2O emissions from fertiliser application at all, claiming that there is little consensus and published data about whether these emissions do in fact occur (Almeida *et al.*, 2014; Dias *et al.*, 2017). Thus, a sensitivity analysis will be employed, using the uncertainty range as provided (0.1-4.6%).

2.1.9.6.3 Indirect N_2O emissions from volatilisation and atmospheric deposition

When N inputs to soil are volatised as ammonia and nitrogen oxides, these gases and their products (NH_4^+ and NO_3^-) can later be redeposited on soils and water bodies (off-site from where the original N was applied). N_2O emissions can then result from these deposited forms of N. N volatilisation and resulting N_2O emissions is only considered for synthetic and organic fertiliser N inputs, and not for N from crop residues, as per EEA 2019 and IPCC 2006/2019 guidance (IPCC, 2006c, 2019; EMEP & EEA, 2019a).

These indirect N_2O emissions from atmospheric deposition (N_2O_{ATD}) can be calculated using IPCC emission factors, which are applied to the fraction of N which has volatilised. IPCC guidelines provide standard values to estimate the fraction of N volatilised as NH_3 and NO_x; however, emissions of NH_3 and NO_x (as NO_2) were modelled in this LCA using EEA guidance (EMEP & EEA, 2019a, 2019b). This allowed for a more accurate estimate of emissions based on fertiliser type and also disaggregation of volatilised emissions between NH_3 and NO_x (whereas IPCC guideline used a simple assumption of all volatilised N as NH_3). Thus, the IPCC emission factor is applied to the amounts of volatised NH_3 and NO_x as calculated using the EEA methodology. Equation 21 thus provides the calculation of indirect N_2O emissions from volatilisation and subsequent atmospheric deposition, adapted from IPCC guidelines (IPCC, 2006c, 2019).

$$E_{N2O,ATD} = ([(E_{NH3,M} + E_{NH3,O}) * (\frac{14}{17})] + [E_{NO2} * (\frac{14}{46})]) * EF_4 * \left(\frac{44}{28}\right)$$

Equation 21 - Indirect N_2O emissions to air from atmospheric deposition

Where,

- $E_{N2O,ATD}$ = the indirect emission of N_2O from atmospheric deposition of the volatilised N fraction, in [kg N_2O].
- $E_{NH3,M}$ = the emission of NH_3 from N in all mineral (synthetic) fertilisers applied for a specific crop, in [kg NH_3], calculated based on EEA 2019 guidance (Equation 8).
- $E_{NH3,O}$ = the emission of NH_3 from N in all organic fertilisers applied for a specific crop, in [kg NH_3], calculated based on EEA 2019 guidance (Equation 9).
- $\left(\frac{14}{17}\right)$ = conversion factor to convert [kg NH_3] to [kg NH_3-N].
- E_{NO2} = the emission of NO_x, modelled as NO_2, from N in all synthetic and organic fertilisers applied for a specific crop, in [kg NO_2], calculated based on EEA 2019 guidance (Equation 10).
- $\left(\frac{14}{46}\right)$ = conversion factor to convert [kg NO_2] to [kg NO_2-N].
- EF_4 = 0.014 (uncertainty range: 0.011-0.017), a constant emission factor for N_2O emissions from atmospheric deposition of N on soils and water surfaces, in [kg N_2O-N (kg NH_3-N + NO_x-N volatilised)$^{-1}$]. This IPCC 2019 emission factor is specified for wet climates only, and thus can be applied to all farms in this study (IPCC, 2019).

2.1.9.6.4 Indirect N_2O emissions from leaching

Leached nitrogen can be transformed through nitrification or denitrification, converting NO_3^- and NH_4^+ to N_2O. This can happen in the groundwater below the land where N additions were applied, in riparian zones receiving runoff or drainage water, or in the above-ground water bodies into which drainage water eventually flows (IPCC, 2019). N inputs that can contribute to these indirect N_2O emissions as a result of leached N include synthetic and organic fertilisers (F_{SN} and F_{ON}) and cover crop residues (F_{CC}).

These indirect N_2O emissions are modelled using IPCC emission factors, applied to the amount of leached N; this is modelled the same for both soil-based and soilless crop cultivation, although amounts of leached N for these processes will differ. Thus, the IPCC emission factors are applied to the calculated values of NO_3-N leached, using the de Willigen 2000 model as a default for soil-based emissions, and a specific emission factor derived from literature for soilless systems. Thus, the IPCC standard value to estimate the fraction of N that is leached as NO_3^- is not used (except when investigating this model in a sensitivity analysis).

The refined IPCC 2019 guidance does not provide any disaggregated Tier 1 emission factors for indirect N_2O emissions from nitrate leaching; thus, the standard Tier 1 emission factor is used. However, differences in nitrate leaching based on fertiliser type and specific farm geography is already accounted for within calculations of nitrate leaching for each crop lifecycle. The calculation for indirect N_2O emissions from nitrate leaching is provided in Equation 21, adapted from IPCC guidelines (IPCC, 2006c, 2019).

$$E_{N2O,L} = E_{NO3} * \left(\frac{14}{62}\right) * EF_5 * \left(\frac{44}{28}\right)$$

Equation 22 - Indirect N_2O emissions to air from N leaching and runoff

Where,

- $E_{N2O,L}$ = the indirect emission of N_2O from nitrate leached or runoff, in [kg N_2O].

- E_{NO3} = the emission of NO_3^- from N in fertilisers and cover crop residues applied to the soil for a specific crop lifecycle, in [kg NO_3^-]. This value is either that as calculated for soil-based crop cultivation (E_{NO3-S}) or soilless crop cultivation (E_{NO3-SL}), as calculated in Equation 11 and Equation 18, respectively. Note that a sensitivity analysis is conducted for soil-based nitrate leaching, applying different leaching emission models (shown in Equation 14, Equation 15, Equation 16, and Equation 17).
- $\left(\frac{14}{62}\right)$ = conversion factor to convert [kg NO_3^-] to [kg NO_3-N].
- EF_5 = 0.011 (uncertainty range: 0.0005-0.025), a constant emission factor for N_2O emissions from N leaching and runoff, in [kg N_2O-N (kg NO_3-N leached/runoff)$^{-1}$].
- $\left(\frac{44}{28}\right)$ = conversion factor to convert [kg N_2O-N] to [kg N_2O].

2.1.9.7 Carbon Dioxide (CO_2) emission calculations

Carbon dioxide (CO_2) emissions to air are considered from applications of urea, limestone (and limestone containing products), and dolomite. Limestone ($CaCO_3$) and dolomite ($Ca(Mg(CO_3)_2)$) are used in agricultural contexts to reduce soil acidity. CO_2 emissions can occur from limestone and dolomite as the carbonate dissolves and releases bicarbonate, which converts into carbon dioxide and water (IPCC, 2006c). In a similar way, urea fertilisation leads to CO_2 emissions. In the presence of water and enzymes, urea ($CO(NH_2)_2$) is converted in the soil into ammonium (NH_4^+), hydroxyl ions (OH^-) and bicarbonate, which then eventually breaks down to CO_2 and water.

These emissions are calculated using IPCC 2006 Tier 1 emission factors (which were not updated in the 2019 refinement), as also applied in ecoinvent v.3.0, WFLDB v.3.0, AGRIBALYSE v.3.0, and Agri-footprint v.4.0. Emission factors for limestone, dolomite, and urea are provided separately and applied to the total mass of each respective input, as defined in Equation 23.

$$E_{CO2} = [(M_{lime} * EF_{lime}) + (M_{dol} * EF_{dol}) + (M_{urea} * EF_{urea})] * \left(\frac{44}{12}\right)$$

Equation 23 – CO_2 emissions from lime, dolomite, and urea

Where,

- M_{lime}, M_{dol}, and M_{urea} = the mass of applied limestone, dolomite, and urea, respectively, in [kg].
- EF_{lime} = 0.12, the emission factor for CO_2-C emissions from limestone application, based on IPCC 2006 guidelines, in [kg CO_2-C (kg applied limestone)$^{-1}$].
- EF_{dol} = 0.13, the emission factor for CO_2-C emissions from dolomite application, based on IPCC 2006 guidelines, in [kg CO_2-C (kg applied dolomite)$^{-1}$]
- EF_{urea} = 0.20, the emission factor for CO_2-C emissions from urea application, based on IPCC 2006 guidelines, in [kg CO_2-C (kg applied urea)$^{-1}$]
- $\left(\frac{44}{12}\right)$ = conversion factor to convert [kg CO_2-C] to [kg CO_2]

2.1.9.8 Phosphorous emission calculations

Phosphorous (P) emissions to water may occur through runoff or leaching. In general, P in soils is relatively immobile, as it tends to precipitate into solid forms. This includes aluminium phosphate in low pH soils and calcium phosphates in high pH soils (Wyatt, Arnall and Ochsner, 2019). However, P can be leached in some soils; this usually happens in sandy soils or those that have high dissolved organic P components, where P inputs are no longer adsorbed to the soil. P can also be transported via runoff to nearby water bodies.

P leaching influences eutrophication impacts; where P exceeds natural inputs by large amounts, this can lead to undesirable growth of phytoplankton. P is often the limiting nutrient for algal growth for inland waters in temperate zones, such as for the countries in this study (Struijs *et al.*, 2011). In this LCA, P emissions to water impact only the freshwater eutrophication pathway during LCIA.

P emissions are modelled separately for soil-based and soilless crop cultivation and are not considered for seedling production (assuming leaching is minimised by growing in containers).

2.1.9.8.1 P emissions from soil-based cultivation

For soil-based crop cultivation, P emissions are modelled based on a constant emission factor as used by Agri-footprint v.4.0 (Durlinger *et al.*, 2017), which estimates that 5% of the P input from organic and synthetic fertilisers reaches freshwater pathways, based on a European-wide study of phosphorus emissions into surface water bodies (Struijs *et al.*, 2011).

Although this is a simple and generalised emission factor, it is applied here because the factor is dependent upon P_2O_5 applied, and thus is linked to management decisions. Conversely, the ecoinvent v.3.0 and WFLDB v.3.0 databases use the SALCA-P model to estimate P leaching and runoff emissions using simple factors of kg P emitted per (area*year), defined for arable versus pasture land uses, with some corrections depending the type of fertiliser applied (mineral fertiliser, manure, or slurry) (Nemecek *et al.*, 2019). It did not seem appropriate to use generalised emission factors based simply on land use, and so this is why the emission factor used in Agri-footprint v.4.0 methodology, based on amounts of applied P, was used.

Thus, the estimation of P emissions is calculated simply using Equation 24.

$$E_{P-S} = F_{P2O5} * 0.4365 * EF_{P,S}$$

Equation 24 – P emissions to water from phosphate fertilisers in soil-based crop cultivation

Where,

- $E_{P\text{-}S}$ = the phosphorous emission to water from fertiliser application in soil-based crop cultivation, in [kg P].
- F_{P2O5}= the mass of phosphate (P_2O_5) in all applied fertilisers, in [kg P_2O_5].
- 0.4365 = conversion factor to convert [kg P_2O_5] to [kg P].
- $EF_{P\text{-}S}$ = 0.05, the emission factor for P emissions from applied P in fertilisers during soil-based cultivation, used within Agri-footprint v.4.0 (Durlinger *et al.*, 2017).

2.1.9.8.2 P emissions from soilless cultivation

Emissions of phosphorus for soilless cultivation were estimated from reported literature values of P in leachate from hydroponic tomato cultivation, as for nitrate emissions. Again, these are only considered for open hydroponic systems.

The percent of P emitted per applied P, as measured in drainage leachate from hydroponic cultivation, was given as 28% for tomatoes grown in perlite (Sanjuan-Delmás *et al.*, 2020) and 39% for tomatoes grown in coco coir (Martin-Gorriz, J.F. Maestre-Valero, *et al.*, 2021). Another value considered was the emission factor advised by Pluimers *et al.* (2000) for glasshouse tomato cultivation in rockwool, which was 10% PO_4-P per applied P. Averaging these three values provides an estimated 26% P leached per applied P, with an uncertainty range of 10-49%, using minimum and maximum values provided in the considered studies. This factor is applied using Equation 25 to calculate P emissions to water for soilless cultivation.

$$E_{P-SL} = F_{P2O5} * 0.4365 * EF_{P,SL}$$

Equation 25 - P emissions to water for soilless crop cultivation (open hydroponic systems)

Where,

- $E_{P\text{-}SL}$ = the phosphorous emission to water for phosphorus in nutrient solution leachate from open hydroponic systems (for soilless cultivation only), in [kg P].
- F_{P2O5}= the mass of phosphate (P_2O_5) in all applied fertilisers in nutrient solution, in [kg P_2O_5].
- 0.4365 = conversion factor to convert [kg P_2O_5] to [kg P].
- $EF_{P\text{-}SL}$ = 0.26 (uncertainty range: 0.10-0.49), a literature-averaged emission factor for P emissions from applied P in fertilisers during soilless tomato cultivation in open hydroponic systems.

2.1.9.9 Heavy metal emission calculations for soil-based cultivation

Heavy metal emissions to water and soil are considered for soil-based crop cultivation phases. These emissions are not modelled for seedlings, since they are grown in containers, and thus emissions to soil/ water are assumed as negligible.

Heavy metal emissions to soil and water are considered for the following elements: Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Nickel (Ni), Zinc (Zn), and Mercury (Hg). This is based on those causing the most environmental concern from agriculture (Nemecek *et al.*, 2019).

Heavy metal emissions are modelled using an adapted version of the SALCA-heavy metal emission model (Freiermuth, 2006), as used in Agri-footprint v.4.0 (Durlinger *et al.*, 2017). A detailed description of this model is included in WFLDB v.3.0 methodology (Nemecek *et al.*, 2019). This LCA includes heavy metal emissions to groundwater through leaching (always positive values) and emissions to agricultural soil (can be positive or negative). Negative values occur when more heavy metals are removed from the soil via leaching and in biomass of exported crop than what is input to the soil. Emissions of heavy metals into surface waters through erosion, although considered by the model, are not included, as insufficient data was available to calculate soil loss. This is consistent with the methodology used in Agri-footprint v.4.0.

The considered inputs of heavy metals to soils include those from synthetic and organic fertiliser application, as well as from atmospheric deposition. Outputs of heavy metals are considered from leaching and removal of biomass from crops. Note that WFLDB v.3.0

methodology (but not Agri-footprint v.4.0) also includes heavy metal inputs from pesticides and seeds. However, this is not included here, as crop seeds are outside the scope of this LCA. For pesticides, the full ingredients are modelled within pesticide processes, and emissions to air, water, and soil from pesticides take into account inert ingredients; thus, it is assumed any relevant heavy metals will be already included within these modelled emissions.

Heavy metal inputs from atmospheric deposition are based on standard values provided by Freiermuth (2006), used within Agri-footprint v.4.0, WFLDB v.3.0, ecoinvent v.3.0, and AGRIBALYSE v.3.0 methodologies. These values have been reproduced in Table 56.

Table 56 *– Atmospheric deposition of heavy metals (Freiermuth, 2006)*

Heavy metal	Cd	Cu	Zn	Pb	Ni	Cr	Hg
Atmospheric deposition [mg $(ha*yr)^{-1}$]	700	2,400	90,400	18,700	5,475	3,650	50

Heavy metal contents of specific fertilisers, both organic and synthetic, were first taken from primary data (supplier information), where this data was available. When this information was not available, heavy metal contents for specific mineral fertilisers (e.g., calcium ammonium nitrate, triple superphosphate, etc.) were estimated from those used in WFLDB v.3.0 guidelines (Nemecek *et al.*, 2019), based on values from Desaules and Studer (1993). Where specific fertilisers were unknown, or where fertiliser mixes were used, average values for N, P, and K mineral fertilisers were used based on Agri-footprint v.4.0 guidelines (Durlinger *et al.*, 2017), with values from Mels, Bisschops and Swart (2008). Heavy metal contents of all composts were estimated from Martínez-Blanco *et al.* (2010), based on an average of home-composting and industrial in-vessel composting with organic food waste and pruning waste feedstock. This excludes only purchased compost in Georgia, for which supplier-specific information was available. Heavy metal contents of manure were based on those used by Agri-footprint v.4.0 (Durlinger *et al.*, 2017); horse manure based on Łapiński and Wiater (2019); and cow manure based on Amlinger, Pollack and Favoino (2004). Heavy metal contents provided by Agri-footprint v.4.0 guidelines for grass, alfalfa, and rapeseed were used to approximate for inputs of hay/straw mulch, alfafameal, and okraseedmeal, respectively (Durlinger *et al.*, 2017).

Heavy metal contents in the biomass of crops and crop residues removed from the farm are accounted for as the heavy metal outputs; values of heavy metals are provided in Table 57. Amounts of heavy metals in tomato fruit are based on an average of those reported in literature (Rossi *et al.*, 2008; Demirbas, 2010; Cherfi, Abdoun and Gaci, 2014; Christou *et al.*, 2014). Heavy metal content of kale leaves (harvested crop) are taken from Korus (2020). The heavy metal contents of above-ground crop residues that are removed from the farm are estimated from mean values in plant material dry matter (averaged from 15 different crops, including mainly grains and legumes) as used in WFLDB v.3.0 methodology (Freiermuth, 2006). These values are then applied to the amounts of above-ground biomass of the kale or tomato plant, based on amounts harvested, as provided in Table 51. Note that this is only counted as a heavy metal output if the crop residue is removed from the farm cropping system. This includes crop residues that are used as animal feed, as well as those composted. For crop residues, cover crop residues, and crop waste that are incorporated into the soil, outputs of heavy metals in biomass are *not* counted since they do not leave the system (Nemecek *et al.*, 2019).

Table 57 *– Heavy metal contents of crops and crop residue biomass*

Heavy metal	**Tomato, organic** [mg (kg fresh matter)$^{-1}$]	**Tomato, conventional** [mg (kg fresh matter)$^{-1}$]	**Kale** [f] [mg (kg fresh matter)$^{-1}$]	**Crop residues** [g] [mg (kg dry matter)$^{-1}$]
Cd	0.033 [a]	0.002 [a]	0.16	0.10
Cu	0.490 [a]	0.46 [a]	11.7	6.60
Pb	0.038 [a]	0.003 [a]	0.10	0.54
Zn	5.90 [b]		14.8	32.0
Ni	0.115 [c]		0.29	1.04
Cr	1.156 [d]		0.27	0.55
Hg	0.003 [e]		0.006 [e]	0.04

[a] Values from Rossi *et al.* (2008), specified for organically-grown and conventionally-grown tomatoes. Values were originally provided in mg heavy metal (kg fresh matter of tomato fruit)$^{-1}$.
[b] Values from (Demirbas, 2010). This was originally given in mg heavy metal (kg dry matter of tomato fruit)$^{-1}$, but was converted to per fresh matter using the 8.2% tomato fruit dry matter value used in this study (Table 51).
[c] Value from Christou *et al.* (2014), given per tomato fruit dry matter, converted to fresh matter as above.
[d] Value from Cherfi, Abdoun and Gaci (2014), given per tomato fruit dry matter, converted to fresh matter as above.
[e] No specific values of Hg in tomato fruit or kale leaves could be found; thus, the generic mean of Hg in plant material (0.04 mg kg^{-1} dry matter), as originally provided by Freiermuth (2006) and used in WFLDB v.3.0 methodology (Nemecek *et al.*, 2019), is used, applying the relevant % dry matter for tomato fruit and kale leaves as in Table 51.
[f] All heavy metal values for kale are taken from Korus (2020), given in mg heavy metal (kg fresh matter kale leaves)$^{-1}$.
[g] All heavy metal values for above-ground biomass from crop residues of kale and tomatoes are approximated from the mean heavy metal values from plant material as originally provided by (Freiermuth, 2006) and used in WFLDB v.3.0 methodology (Nemecek *et al.*, 2019). These values are applied to the amounts of above-ground biomass, as related to harvested crop, estimated in Table 51.

Leached heavy metals are also accounted for as heavy metal outputs and are modelled as emissions to groundwater. This is estimated based on standard values as used in WFLDB v.3.0 and Agri-footprint v.4.0 methodologies, originally from Wolfensberger and Dinkel (1997). Estimated heavy metal leaching values per area per year are given in Table 58.

Table 58 *– Leaching of heavy metals (Wolfensberger and Dinkel, 1997)*

Heavy metal	**Cd**	**Cu**	**Zn**	**Pb**	**Ni**	**Cr**	**Hg**
Leached amount (m_{leach}) [mg (ha*yr)$^{-1}$]	50	3,600	33,000	600	n/a	21,200	1.3

These leaching amounts are then applied to specific crop lifecycles by multiplying by the cropped area used (ha) and the total time that area is used for that crop out of the year. However, heavy metal leaching must be allocated specifically to agricultural activities using an allocation factor. This is because not all heavy metal accumulation is caused by agricultural production; there is also atmospheric deposition of heavy metals from other activities in the area. The allocation factor can thus be calculated as in Equation 26.

$$A_i = \left(\frac{M_{ag,i}}{M_{ag,i} + M_{dep,i}}\right)$$

Equation 26 – Allocation factor for heavy metal emissions from agricultural activities

Where,

- A_i = allocation factor for the share of emissions of heavy metal *i* from agricultural activities, out of total inputs of heavy metal *i*. This is a dimensionless value.
- $M_{ag,i}$ = the input of heavy metal *i* from agricultural inputs, including synthetic fertilisers, organic fertilisers, and bought-in mulches (e.g., hay), in [mg heavy metal *i* $(ha*yr)^{-1}$].
- $M_{dep,i}$ = the input of heavy metal *i* from atmospheric deposition, as provided in Table 56, in [mg heavy metal *i* $(ha*yr)^{-1}$].

This allocation factor can then be used in conjunction with the values in Table 58 (represented as m_{leach}) to calculate leached heavy metal emissions to groundwater, using Equation 27.

$$M_{leach,i} = m_{leach,i} * A_i$$

Equation 27 – Leached heavy metal emissions to water for soil-based crop cultivation

Where,

- $M_{leach,i}$ = the output of heavy metal *i* from the amount that has been leached from the soil, attributed to agricultural activities, in [mg heavy metal *i* $(ha*yr)^{-1}$].
- $m_{leach,i}$ = the average amount of leached heavy metal emission *i* as per **Table 58**, in [mg heavy metal *i* $(ha*yr)^{-1}$].
- A_i = the allocation factor to allocate emissions of heavy metal *i* to only agricultural activities, calculated in Equation 26. This is a dimensionless value.

Heavy metal emissions to soil are then calculated based upon the net flow of heavy metals into and out of the soil. This thus includes all heavy metal inputs from fertilisers and atmospheric deposition minus the outputs from crops and crop residue removed from the soil and from heavy metal leaching, all multiplied by the allocation factor (A_i). In this way, the final emission to soil (E_{HM-S}) for each heavy metal (*i*) can be calculated using Equation 28.

$$E_{HM-S,i} = (M_{ag,i} + M_{dep,i} - M_{crop,i} - M_{leach,i}) * A_i$$

Equation 28 – Heavy metal emissions to soil from soil-based crop cultivation

Where,

- $E_{HM-S,i}$ = the emission to soil for heavy metal *i* due to agricultural activities, in [mg heavy metal *i* $(ha*yr)^{-1}$].

- $M_{ag,i}$ = the input of heavy metal *i* from all agricultural inputs, including synthetic fertilisers, organic fertilisers, and bought-in mulches (e.g., hay), in [mg heavy metal $(ha*yr)^{-1}$]
- $M_{dep,i}$ = the input of heavy metal *i* from atmospheric deposition, as provided in Table 56, in [mg heavy metal $(ha*yr)^{-1}$].
- $M_{crop,i}$ = the output of heavy metal *i* from the amount in all crop and crop residues which are removed from the soil [mg heavy metal $(ha*yr)^{-1}$]. Crop residues that are later incorporated back into the soil are not considered here.
- $M_{leach,i}$ = the output of heavy metal *i* from the amount that has been leached from the soil, as calculated in Equation 27, in [mg heavy metal *i* $(ha*yr)^{-1}$].
- A_i = the allocation factor to allocate emissions of heavy metal *i* to only agricultural activities, calculated in Equation 26. This is a dimensionless value.

2.1.9.10 Heavy metal emission calculations for soilless cultivation

Heavy metal emissions to water (in leachate) are considered for soilless cultivation, again only for open hydroponic systems, as closed systems are assumed to produce zero leachate. The particular heavy metals modelled are the same as for soil-based cultivation (Cd, Cr, Cu, Pb, Ni, Zn, and Hg), based on those most relevant to agricultural-related environmental impacts. Heavy metal emissions to soil are not considered for hydroponic tomato cultivation, in accordance with AGRIBALYSE v.3.0 processes and methodology (ADEME, 2020; Koch and Thibault, 2020).

There are no specific models developed to calculate heavy metal emissions from hydroponic systems, so this has been modelled as a simpler rendition of the SALCA-heavy metal emission calculation used to approximate heavy metal emissions to soil for soil-based cultivation (Equation 28). Since this is considered for hydroponic cultivation, all emissions are assumed as emissions to water. Further, since the model is for an isolated, indoor soilless cultivation system, one does not have to consider atmospheric deposition of heavy metals and thus also does not need to consider allocation of the heavy metal input. Applying these assumptions to Equation 28, the result is a simple net balance calculation of heavy metal input in the fertilisers minus heavy metal output in the crop. It is assumed that all resulting excess heavy metals will be contained in the drained leachate and will thus be emitted to water.

The heavy metal input is calculated based on the heavy metal content of each macro- and micronutrient fertiliser. As for the soil-based cultivation calculation, heavy metal contents for common NPK fertilisers are approximated from values given in WFLDB v.3.0 guidelines (Nemecek *et al.*, 2019), based on values by Desaules and Studer (1993). The heavy metal output in the crop is calculated based on the total crop biomass, since for the particular soilless system of interest in this study, all crop biomass is removed from the system. This is approximated using heavy metal contents of the crop and crop residues as given in Table 57; amounts of residues per crop biomass is estimated using the relevant ratios in Table 51.

Based on these inputs and outputs, the emission of a particular heavy metal *i* ($E_{HM-SL,i}$) is calculated using Equation 29.

$$E_{HM-SL,i} = M_{ag,i} - M_{crop,i}$$

Equation 29 - Leached heavy metal emissions for soilless crop cultivation (open hydroponic systems)

Where,

- $E_{HM\text{-}SL,i}$ = the emission to water for heavy metal *i*, from amounts in the nutrient solution leachate in open hydroponic systems (for soilless cultivation only), in [mg heavy metal *i* $(ha*yr)^{-1}$].
- $M_{ag,i}$ = the input of heavy metal *i* from all fertiliser inputs, in [mg heavy metal $(ha*yr)^{-1}$].
- $M_{crop,i}$ = the output of heavy metal *i* from the amount in all crop and crop residues which are removed from the system [mg heavy metal $(ha*yr)^{-1}$]. For the soilless system of interest in this LCA, this includes total crop biomass (harvest, harvest waste, and above- and below-ground residues).

There is some uncertainty associated with this calculation, especially since heavy metal uptake in the crop is based on rough approximations on amounts of crop residues and uses generalised heavy metal contents for the average plant residue, which is not crop-specific (Freiermuth, 2006; Nemecek *et al.*, 2019). However, there was little literature data available on heavy metal contents of leachate following hydroponic tomato cultivation. For an open hydroponic system growing tomatoes in perlite, Sanjuan-Delmás *et al.* (2020) found that 63% of input iron (Fe) was leached and 34% of input zinc (Zn) was leached. The above method alternatively calculated 65% of Zn leached in an open hydroponic system, based on the fertiliser inputs for hydroponic tomato production as provided in Appendix A, Table 72. Thus, this method may be over-estimating heavy metal emissions; however, due to a lack of other relevant models or emission factors for soilless systems, Equation 29 will be used as a basic estimate.

2.1.9.11 Pesticide emissions

Pesticide emissions are modelled similarly for all seedling and crop processes. It is based on methodology employed in Agri-footprint v.4.0 processes, which applies 9% of the active ingredient amount in a pesticide product as an emission to air; 1% to water; and 90% to soil (Durlinger *et al.*, 2017). These same emissions were also applied to the amount of inert ingredients in pesticide products, as listed on material safety data sheets for the specific products used by farmers in the study. Although inert ingredients, or co-formulants, are not typically modelled in LCAs, this has been done because inert ingredients can also have similarly adverse effects on human health and the environment as do active ingredients (Cox and Surgan, 2006; Eddleston *et al.*, 2012; Mesnage and Antoniou, 2018; Mesnage, Benbrook and Antoniou, 2019). Because data was gathered on specific pesticide products, hazardous inert ingredients could be found on safety data sheets for each pesticide product, and where these were listed, they were modelled.

For biological pesticides that contain live strains of fungi or bacteria, it is assumed that there is a 100% emission to soil.

2.1.9.11.1 *Uncertainty for soilless cultivation and glasshouse production*

Greenhouses or glasshouses influence pesticide transfer by generally decreasing their spread at the regional level. There is no standard method for estimating pesticide emissions in glasshouses, and indeed, many greenhouse-based agricultural LCAs do not take into account pesticide emissions at all (Boulard *et al.*, 2011; Torres Pineda *et al.*, 2021). In this LCA, the calculation pesticide emissions for soilless cultivation in glasshouses will follow that as for soil-based cultivation. This choice is based on a review of methodologies currently employed in literature.

Although pesticide emissions to the surrounding environment differ within greenhouses compared to that in open-air soil-based systems, the decision to approximate emissions in the same way for both cases was based largely on an assessment by Boulard *et al.* (2011). In comparing impacts of pesticide application in hydroponic tomato glasshouses, Boulard *et al.* (2011) assessed three scenarios of calculating pesticide emissions: 1) using a model developed by Hauschild (2000) and adapted by Antón *et al.* (2004) to estimate emissions leaving the greenhouse; 2) assuming 100% of active ingredients are emitted to soil; and 3) assuming 100% of active ingredients are emitted to air.

In the first scenario, Antón *et al.* (2004)'s adapted model attempts to estimate the amount of active ingredient that is emitted from the greenhouse into the surrounding environment, taking into account several factors including: loss via leaching, loss via drift, and loss via volatilisation from the soil and from the plant, based on the leaf area index. However, there is a high degree of uncertainty with this method, as it depends on specific climatic conditions on the day of spraying and daily evaporative rates of active ingredients from the plant and from the soil (Antón *et al.*, 2004; Boulard *et al.*, 2011). The second scenario follows ecoinvent and WFLDB crop cultivation processes; these crop processes attribute 100% of the pesticide active ingredient as an emission to soil for both soilless and soil-based farming systems (Nemecek and Schnetzer, 2011; Quantis and Agroscope, 2019; ecoinvent Centre, 2021).

Comparing these scenarios for the hydroponic production of tomatoes in a multi-span glasshouse, Boulard *et al.* (2011) found that differences in ecotoxicity impacts were negligible, as pesticide use is minimal in this production system already. Additionally, the contribution of pesticide application to ecotoxicity categories, in comparison to other inventory processes, was negligible overall (≤0.36%).

Since the only farm in this study employing soilless cultivation (UK conventional tomato production) also uses a multi-span glasshouse with only minimal biopesticide use, it can be assumed that the method of calculating pesticide emissions will not have a significant influence on final results. Thus, the simple method of attributing the same emissions as for soil-based systems is used. This method will also be employed for any pesticide application during seedling production in nurseries that use glasshouses and do not grow in the soil.

2.2 Soil health and ecosystem services from local farms in the UK

To expand upon the outcomes of the LCA study, a soil health assessment was then performed on a subset of the case study farms located in England, UK. The aim of this study is to uncover other ecosystem services and disservices that may be provisioned from these farms, but are not captured through the LCA. A selection of soil health metrics from the Soil Health Institute's Tier 1 list were chosen to serve as indicators for important ecosystem services (Soil Health Institute, 2017), which align with current UK policy objectives (see: Section 1.3.2). These indicators include: soil organic carbon; bulk density; water-holding capacity; macronutrient concentrations, including N, P, and K; C/N ratio; micronutrient concentrations; and heavy / trace metal concentrations.

2.2.1 Soil sampling locations

In October-November 2021, soil samples were collected from 10 of the 14 UK farms included in the LCA study to test for indicators of soil quality and ecosystem services. Those not included were unavailable for soil sampling at the time of the study; additionally, the hydroponic tomato producer was not included as soil is not used for this production system. Table 59 provides an overview of the main characteristics of each farm site included in the study and the areas sampled on each farm. Nomenclature of the farms is consistent with that used within the LCA study (see: Section 2.1.2.3); 'U', 'PU', and 'R' designate urban, peri-urban, and rural farms, respectively, while 'O' and 'C' indicate organic or conventional farms, respectively. Definitions of urbanisation for each farm were based on those applied for lifecycle assessment. Urban and peri-urban farms are collectively grouped in this study as 'urbanised', whilst all other farms are considered rural.

As also explained in Section 2.1.2.3, organic farms are considered to be all those following organic principles, even if not officially certified as organic. In this study, organic farms are characterised by the restricted use of synthetic fertilisers and pesticides; composts, manures, and cover crops are thus the main fertility sources for these farms, as detailed by farmers during the LCA study. One farm (R-C-5) is considered as mainly conventional; however, they do have a small portion of organically-certified fields, one of which was sampled for inclusion as an organically cultivated area. It should be noted that this field was only in its first year of official organic certification after two years of transition, and previously had been used for conventional arable or horticultural cultivation. In contrast, all other organic farms have been cultivating under organic principles for at least 10 years prior to sampling in 2021. The amount of time that each farm site has been under its current management is provided in Table 59 ('years of management'), in reference to the time of soil sampling (2021). It should also be noted that all conventional farms rent at least a portion of their land, and that fields sampled were likely to have been previously managed for arable, horticulture, or livestock purposes before the establishment of these farms (particularly for R-C-3 and R-C-5).

Specific locations of each farm site are not provided in order to maintain confidentiality of the farms; however, information about the region, soil type, and climate on each farm is provided in Table 59. Regional designations are based on UK climate district map, provided by the UK Met Office (UK Met Office, 2022b). The annual mean, minimum, and maximum temperature, sunshine duration, and rainfall are based on the figures provided by the UK Met Office for these regions in 2021 (UK Met Office, 2021), the year that soil samples were taken. Elevation ranges provided for each farm is based on elevation for each soil sampling site, determined using Google Earth Pro v.7.3.6 (Google, 2022). Soil type for each sampled farm area was determined using the World Reference Base for Soil Resources (WRB) maps developed for England by the National Soil Resources Institute at Cranfield University (Cranfield University, 2006), viewed through the UK Soil Observatory (UKSO) map viewer (https://mapapps2.bgs.ac.uk/ukso/home.html). Listed soil types follow a harmonised global nomenclature (Cranfield University, 2023). Soil parent material was similarly determined using the UKSO map viewer, based on maps provided by the British Geological Survey (British Geological Survey, 2001); these use the European Soil Bureau descriptions for soil parent material types (European Soil Bureau, 2001). In some cases, soil types and soil parent materials varied across sampling locations on each farm, in which case several types for the same farm will be listed in Table 59, although mostly these were the same.

Soil samples were collected from different landscape areas on each farm to provide a representative overview of ecosystem service provisioning across the whole farm site. The areas sampled on each farm are designated in Table 59. These include the following areas, when they were present on or near the farm: cultivated vegetable fields (F); polytunnels (P); fields sown with herbal leys within a crop rotation (HL); grassy areas not used for crop production, including grassy field margins, permanent pasture areas, and unmanaged grassy fields (G); hedgerows (H); orchards (O); and woodlands (W). Different landscape areas on each farm were identified during farm visits, based on guidance from the farmer. Not all landscape areas existed on each farm, so in some cases, only certain areas were sampled. However, at least one field or polytunnel used for vegetable cultivation was always sampled, selecting those that the farmer identified as an 'average' field. Additionally, in some cases, soil samples were taken from land areas adjacent to the farm to provide a sample for a particular landscape, if that landscape did not exist on the farm; this is the case for the woodlands sampled for PU-O-4 and R-C-4. Finally, in some cases multiple fields, polytunnels, woodlands, or grassy areas were sampled to provide a more representative depiction of these areas across the farm; for example, samples were taken from older and younger woodlands and cultivated plots for PU-O-3 and R-O-1, and from no-till and tilled fields for U-O-2.

For statistical analysis, samples from different landscapes were grouped based on whether they were from uncultivated land areas, organically-managed areas, or conventionally-managed fields. The prior case refers to land area on or near farms that are not used for food production or are unmanaged, and thus do not receive any fertiliser or pesticide inputs, including: woodlands, hedgerows, unmanaged grassy fields, and grassy field margins. Organic land refers to land used for food production, cultivated in line with organic principles. This includes areas found on organic farms such as: cultivated fields, cultivated soil in polytunnels, pastures, and orchards. Similarly, conventional land refers to land cultivated using non-organic, or conventional, principles. On conventional farms, this includes cultivated fields and pastures; no conventional farms included in the study had polytunnels or orchards.

Table 59 *– Overview of farm sites included in soil health assessment and soil sampling locations*

	U-O-2	**PU-O-1**	**PU-O-2**	**PU-O-3**	**PU-O-4**	**R-O-1**	**R-O-2**	**R-C-3**	**R-C-4**	**R-C-5**
Farm management	Organic	Organic	Organic	Organic	Organic	Organic	Organic	Conventional	Conventional	Conventional / Organic
Urbanisation	Urbanised	Urbanised	Urbanised	Urbanised	Urbanised	Rural	Rural	Rural	Rural	Rural
Farm size (acres)	12	5	10	15	10	5	31	400	6,500	2,080
Years of operation	10	12	17	21	25	25	20	8	>200	22; 1 for organic
Landscapes sampled	F (2), P, HL, H, W n=6	F, P, HL, G (margin), H, O, W n=7	F, P, HL, G (margin), H, O, W n=7	F (2), G (unmanaged), G (pasture) H, O, W (2) n=8	F, P, HL, H, O, W n=6	P (2), G (unmanaged), H, O, W n=6	F, P, HL, G (margin), H, O, W n=7	F, HL, G (pasture), H, W n=5	F, HL, G (pasture), H, W n=5	F (conventional), F (organic), HL (conventional), G (margin), W n=5
Region	SW	E & NE	E & NE	E & NE	E & NE	NW	Central S	NW	SW	E Anglia
Elevation (m)	40-43	167-175	198-207	70-90	15-18	9	148-156	27-35	90-103	3-4
Soil type	Luvisol	Planasol	Cambisol	Stagnosol	Stagnosol	Gleysol	Cambisol	Planasol	Cambisol / umbrisol	Gleysol
Soil parent material	Sandstone	Mudstone and sandstone	Sandstone	Glacial till	Glacial till	Riverine clay and floodplain sands and gravel	Siltstone and limestone	Glacial till	Basalt; mudstone & sandstone; argillite-slate	Quaternary marine/ estuarine clay/silt
Minimum annual temperature (°C)	6.62	5.64	5.64	5.64	5.64	5.86	6.60	5.86	6.62	6.51
Mean annual temperature (°C)	10.25	9.40	9.40	9.40	9.40	9.37	10.59	9.37	10.25	10.46
Maximum annual temperature (°C)	13.90	13.17	13.17	13.17	13.17	12.90	14.59	12.90	13.90	14.42
Annual sunshine duration (hours)	1,504	1,508	1,508	1,508	1,508	1,443	1,494	1,443	1,504	1,502
Annual rainfall (mm)	1,275	800	800	800	800	1,390	806	1,390	1,275	637

2.2.2 Soil sampling strategy

Within each landscape area of interest on each farm, a sampling location was selected which appeared to be representative of the area as a whole. Samples were not collected from the edges of the area of interest, unless the edge was an area of interest itself (e.g., grassy field margins). A central point was identified in each sampling location, and then soil samples were collected around that central sample point in a 1mx1m square area. Coordinates of each sample location was recorded in Google Maps, although this information was for use of the researcher and is not shared to maintain confidentiality of the farms. An Eijkelkamp bulk density corer was used to extract soil samples at two depth increments (0-10cm and 10-20cm) (Dobson *et al.*, 2021). Within each testing area, four soil samples were taken at each depth and considered as ‘pseudo-replicates’ to derive a site mean (Edmondson *et al.*, 2011), giving a total of eight samples taken within each 1mx1m quadrat. Overall, 496 soil samples were taken across 62 landscapes on 10 farms. After sampling, each individual soil core sample was dried at 105 °C for 48 hours and weighed.

2.2.3 Soil processing and analysis

Water-holding capacity (WHC) was measured on one out of four ‘pseudo-replicate’ soil samples that were taken from each depth and each quadrat (n=124). These samples were passed through a 9mm sieve prior to analysis. WHC was performed using the ‘European’ maximum water-holding capacity method of Gardner (1986). This a gravimetric method that involves saturating the soil sample with water in a cylinder, placed on an absorbent membrane to draw away excess water; in this case, grade 42 Whatman ashless filters were used. After reaching a point of equilibrium, the sample is weighed so that water weight can be determined. WHC is then calculated as the weight of the water held in the sample vs. the sample dry weight, expressed as a percentage.

Two out of four soil samples from each depth and each quadrat (n=248) were then homogenised into a fine powder using an agate ball-mill (Pulverisette; Fritsch, Idar-Oberstein Germany). Each ball-milled sample was then passed through a 2mm sieve to remove any stones or debris that were still present after milling; samples were then redried at 105°C. Material >2 mm was retained, weighed, and subtracted from the original soil dry weight. This material was then volumetrically measured using water displacement and the volume was subtracted from the total volume to calculate soil bulk density (BD) as the soil weight per volume (g cm^{-3}) (Dobson *et al.*, 2021).

After BD determination, inorganic carbon was removed from 90mg of soil from each sample using 6 M HCl. Samples were re-dried at 105°C for 24 hours and re-weighed; the remaining soil was then analysed for soil organic carbon (SOC), nitrogen (N), and sulphur (S) in an elemental analyser (Vario MICRO Cube; Elementar, Germany). SOC, N, and S concentrations as output from the instrument, in g (g acid-treated soil)$^{-1}$, were then adjusted to be representative of the original dry soil weight by multiplying by the weight of the acid-treated soil and re-dividing by the original soil weight.

A portion of each ball-milled soil sample (n=248) was also used to test the total concentration of heavy metals and other macro- and micro-nutrients via inductively coupled plasma mass spectrometry (ICP-MS), as per Crispo et al. (2021). First, two of the ‘pseudo-replicate’ samples taken from the same sample quadrat at the same depth were combined at equal weights to represent one sample, resulting in a total of n=124 samples for further analysis. Aqua regia digestion was then performed on these samples in accordance with ISO 11466:1995 (ISO, 1995). For this process, 0.25 g of each combined soil sample was mixed with 2ml HNO_3 (65–67 %) and 6 ml HCl (34-37 %) in 50 ml glass tubes and then allowed to stand for 16 h at room temperature. After, samples were digested for 2 h at 120°C on a heating block. Once cooled, the digested samples were filtered using grade 42 Whatman ashless filters and diluted by 10x with ultra-pure water. ICP-MS was then performed on the digested samples to measure the total soil concentration of 18 soil nutrients and heavy metals (Agilent 7500ce; Agilent Technologies, CA, USA). This included the following elements: Boron (B), Calcium (Ca), Phosphorus (P), Potassium (K), Magnesium (Mg), Molybdenum (Mo), Manganese (Mn), Selenium (Se), Arsenic (As), Cadmium (Cd), Chromium (Cr), Cobalt (Co), Copper (Cu), Iron (Fe), Nickel (Ni), Lead (Pb), Mercury (Hg), and Zinc (Zn).

Quality assurance of the elemental analysis following aqua regia digestion was ensured through the inclusion of reagent blanks, use of analytical reagent grade chemicals, and internal reference samples for the ICP-MS (Crispo *et al.*, 2021). All glassware was soaked in 1% HCl for 24 h and rinsed with ultra-pure water. Additionally, for quality assurance of trace metal analysis in particular, certified soil reference samples of were included within the digestion and analysis (European Reference Materials: ERM-CC141, loam soil). Certified values for the aqua regia extractable concentration of As, Cd, Co, Cr, Cu, Hg, Mn, Ni, Pb, and Zn, as well as the measured values of these elements within this study, are provided for the reference sample in Table 60. This table also shows the percent recovery of these elements within the study analysis, calculated as the measured concentration for reference samples as a percentage of the certified concentration values.

Recovery was adequate for Pb, Ni, Hg, and Zn, ranging from 84-111%, but low for all other trace elements. For As, Cr, Co, Cu, and Mn in particular, recovery ranged from 37-56%; thus, it is expected that the actual concentrations of these elements within the other soil samples would be approximately 2-3x of those measured and reported in this study. Even though recovery of these elements was low, these measurements were still retained as it is assumed that this would not affect comparative analyses between sample groups. Limits of detection from the ICP-MS are further provided in Appendix E, Table 79 for the main elements that are presented within the results of Chapter 4.

Table 60 *- Recovery of aqua regia soluble trace metals from soil reference samples*

	As	**Cd**	**Cr**	**Co**	**Cu**	**Pb**	**Mn**	**Hg**	**Ni**	**Zn**
Certified value (mg kg^{-1})	7.5 ± 1.4	0.25 ± 0.04	31 ± 4.0	7.9 ± 0.9	12.4 ± 0.9	32.2 ± 1.4	387 ± 17	0.08 ± 0.008	21.9 ± 1.6	50 ± 4.0
Measured value (mg kg^{-1})	3.4	0.2	11.6	4.5	5.9	29.5	217.8	0.1	18.4	46.8
Recovery (%)	45%	74%	37%	57%	48%	92%	56%	111%	84%	94%

2.2.4 SOC storage per area

SOC density (mg cm^{-3}), used to determine SOC storage per area, was calculated using SOC concentration (mg g^{-1} soil) and soil bulk density (g cm^{-3}) *without* the subtraction of the volume >2mm (Edmondson *et al.*, 2012). Thus, this bulk density measure excludes the weight of >2mm material but includes the volume of this material to provide a more accurate representation of C storage in the *soil* for any given sample volume. SOC storage per area (kg m^{-2}) was then calculated using SOC density, considering storage to 10cm and 20cm depths.

2.2.5 Statistical analysis

Prior to statistical analysis, sample data was averaged for each farm landscape with the same management type (organic, conventional, and uncultivated) to avoid any potential pseudo-replication. In particular, the bulk density, C, N, and S values measured for the two replicate samples from each farm landscape analysed were first averaged. Additionally, when two areas of the same landscape *and* management type were measured on the same farm (e.g., two woodlands or two fields), measured parameters were also averaged between these samples. Statistical analysis was then performed considering each landscape type on each farm as one sample variable (n=58).

2.2.5.1 Principal component analysis

Principal component analysis (PCA) is an unsupervised multivariate statistical method, in such that it can be used to indicate general hypotheses from the data without any prior hypotheses or other data classifications. This type of exploratory analysis allows one to visualise complex datasets in a simpler way, observe interrelationships between samples and variables in the data, and identify patterns (Sena *et al.*, 2002).

PCA reduces high-dimensional data to a more manageable set of new variables, called principal components (PCs). PCs are linear combinations of the original variables in the dataset that explain the most variance in the data, extracted in order of descending variance (i.e., the first PC will explain the maximum variance in the data). The amount of PC present within a particular *sample* is the ‘score’, and the contribution that each original *variable* makes to a PC is known as the ‘loading’ (Ivosev, Burton and Bonner, 2008).

Plotting the scores of the first PCs gives a visualisation of the data within the principal component space, which allows for relationships between samples to be observed; in particular, clusters or groups of data may emerge. Clustered PC scores mean that there are groups of data points that are more similar to each other than the data points in other clusters. Differentially labelling these scores based on identifying variables can highlight the characteristics that may be distinguishing these groups.

On the other hand, plotting the loadings can be useful to understand how the original variable contributes to the PCs, thus allowing for the identification of the variables that appear to be driving the most variance in data. Loading plots also help to deduce information about the correlation of different variables in the data and identify groups of variables that may be driving similar variance, which is useful for data reduction. The size of the loading vector indicates the importance of a variable within the PC model. The correlation of variables is related to the cosin of the angle between loading vectors; thus, the smaller the angle between vectors, the higher the correlation between features (Ivosev, Burton and Bonner, 2008) .

Perpendicular variables are uncorrelated, and vectors in opposite directions (e.g., 180° apart) are negatively correlated.

Biplots represents PC scores and loadings on the same plot, allowing for the relationships between variables and samples to be visualised (Gabriel, 1971). If a variable loading is close to an object, it will have a high direct influence on it. Thus, biplots allow for the variables with the highest loadings in the direction of each object or cluster to be identified; In this way, PCA can be used for data reduction by identifying fewer variables that explain the majority of variation in the dataset; this helps to inform further statistical analysis within high-dimensional datasets.

Soil data is highly dimensional as many factors can drive variance between samples, such as soil type, climate, sampling date, location, management, landscape, sample depth, and vegetation types, among others (Davis *et al.*, 2009). Thus, PCA can be used to simultaneously study several soil properties as a function of these many experimental and geographical features, and then can be used for data reduction by identifying a minimum dataset (Sena *et al.*, 2002; Davis *et al.*, 2009; Mann *et al.*, 2019). In this way, PCA has been used to identify correlations between various soil properties, and thus help inform, identify, and select soil heath indicators (Andrews, Karlen and Mitchell, 2002; Mukherjee and Lal, 2014; Fine, van Es and Schindelbeck, 2017; Yu *et al.*, 2018; Rinot *et al.*, 2019).

In this study, PCA was used to reduce the multidimensional dataset because of the large numbers of samples and numerous elements and soil properties measured for each sample. PCA was performed using GraphPad Prism v.9.5.0 for Windows (GraphPad Software, 2022), and all PCA figures were also generated using this software.

PC loadings were used to identify the main variables contributing to variance in the data; this was particularly important to identify which elemental concentrations to select for investigation in further statistical analyses. Additionally, PC scores were analysed for clustering by labelling scores based on different sample characteristic data as previously provided in Table 59, including: soil type; soil parent material; regional area; minimum, average, and maximum annual temperature; annual rainfall; annual sunshine duration; farm size; management type (organic, conventional, or uncultivated land); and urbanisation (urban vs. rural areas). Those that resulted in the most distinct clustering were again selected as the grouping parameters for further statistical analysis.

2.2.5.2 T-tests and ANOVAs

Descriptive statistics were generated for the measured soil parameters and grouping variables identified to be of interest from the PCA. Mean, median, interquartile range, and outliers are provided using box plots, which were produced in Origin(Pro) v.2022b (OriginLab, 2022). Tests of statistical significance were also performed for these groupings and designated on the box plots.

T-tests and ANOVAs (analysis of variance) were performed using GraphPad Prism v.9.5.0 (GraphPad Software, 2022). T-tests were carried out to determine significant differences between the means of urban (n=31) and rural (n=27) samples, whilst ANOVAs were performed to determine significant differences between the means of sample data from uncultivated (n=25), organically-cultivated (n=25), and conventionally-cultivated land areas (n=8). Data was transformed where necessary to qualify for assumptions of normality using

Shapiro-Wilks tests ($p \geq 0.5$) and homogeneity of variances, using the F test when performing T tests and Bartlett's test when performing ANOVAs ($p \geq 0.5$). However, in cases that these assumptions were not met despite data transformation, non-parametric tests were performed. These included the Mann-Whitney U test as a nonparametric option to the T test and the Kruskal-Wallis test as a nonparametric option to ANOVAs. Mean separation was evaluated using the Tukey post-hoc test for parametric data and Dunn's multiple comparison test for non-parametric data. Significance was assumed based on a $p < 0.05$. The statistical test used, and any types of data transformation performed, are designated within the results for each test variable.

2.3 Experiences of food insecurity and community responses during COVID-19: A case study in the UK

The final research project of this thesis focused on food security as a key aspect of food system sustainability. The aim of this study was two-fold. For one, this project aimed to capture the unique and individual experience of food insecurity and its main drivers. This was investigated during the COVID-19 crisis, which was particularly applicable to this aim, as many were experiencing new barriers to accessing food (Loopstra, 2020). Secondly, this study aimed to explore the role of community food responses during this time of crisis.

To achieve these aims, a collaborative research project was initiated with a local community food organisation in Sheffield, England, called Foodhall (legal name, Project Foodhall C.I.C). Foodhall is a grassroots, community-run food project located in the city centre of Sheffield. The project soon became one of the largest emergency food responses in the city. Because the service was free and open to everyone living in Sheffield, this served as an ideal case study to explore issues of food access across the Sheffield population. In addition, the author also served as a volunteer at Foodhall during their COVID-19 emergency food response, and therefore was able to capture data and stories with good knowledge of the organisation and its operations.

This project employed primarily a qualitative methodology, using interviews the capture the lived experiences of people utilizing Foodhall's food support services. However, quantitative data was also gathered from Foodhall volunteers and analysed by the author to understand the scope of the project's food response.

A further goal of the project, beyond this thesis, was to use the critical and timely data gathered during this research to inform COVID-19 crisis response efforts and related food policy. Thus, a major component of the project was communicating results to community food projects across the country; the local city council; and the national government. This research thus provided essential information about the barriers to food access and necessary services that occurred as a result of the pandemic.

2.3.1 Case study background: the 'Foodhall Project'

The Foodhall Project (hereafter referred to as 'Foodhall') is a community kitchen and café in Sheffield, England that is "managed by the community, for the community" (https://www.foodhallproject.org/). Founded in 2015, Foodhall provides hot meals, cooked largely from surplus food, in a communal setting on a 'contribute-what -you-can' basis. As a social eating space, the project aims to bring people together around food to tackle the combined issues of food insecurity, social isolation, social inequality, and food waste (The Foodhall Project, 2020). The café serves as a space for everyone to share food, drink, company, skills and time. In addition to the café space, Foodhall also acts as a community hub for art and culture, hosting various live music events, talks, performances, and art workshops, with a focus on bringing a diverse range of people together. Its physical location in the city centre of Sheffield, situated between some of the most and some of the least deprived postcodes in the country, provides an opportunity for the project to help bring all people together and bridge this divide.

Foodhall’s culture is founded on an asset-based community development ideology (Mathie and Cunningham, 2003), aiming to provide opportunities for all people to actively engage with the project and with each other. Foodhall operates a non-hierarchal, flat management structure. They employ a small staff team to organize the project and manage finances, but ultimately the project is operated by a suite of volunteers, all of whom have a responsibility and a say in how the project is managed. Foodhall provides opportunities for everyone who visits to actively engage with the project, aiming to blur the line between the traditional ‘service provider’ and ‘service user.’ In this way, the project promotes active citizenship and a sense of responsibility by everyone involved.

Volunteers and part-time staff alike to are encouraged to create new projects and initiatives within the community hub. Branch projects are thus started by those who wish to bring something new to the space and share their skills and interests with others. Examples of branch projects include art and music workshops, bike repair classes, putting on gigs and art exhibitions, a community radio, food preservation and fermentation, foraging, gardening, and more. The flexibility and openness of the project is key to people feeling comfortable in the space and getting people from a wide range of backgrounds involved.

As lockdown restrictions were imposed by the UK government on 23 March 2020 following the rise of the coronavirus pandemic, Foodhall was unable to operate their traditional café and event space. Foodhall thus responded by organizing an emergency food distribution effort, which included free parcel delivery as well as a front-of-house take-away service. This study aims to capture the scope of this emergency response effort (with quantitative data collected and provided by the project), as well as the stories of those struggling to access food, as provided through qualitative interviews with food parcel recipients.

2.3.2 Positionality statement for research with Foodhall

This section extends upon the positionality statement provided at the start of this thesis (see: Section 1.8) to provide further context relevant to this particular research study. The background for me engaging in this research was through my volunteer role at Foodhall. Throughout the first COVID lockdown, I consistently volunteered in Foodhall’s warehouse, packing food parcels for distribution.

This role as a volunteer facilitated access to research with the organisation and also with food parcel recipients. I was able to work with other volunteers and contribute to the impact data tracking that provided quantitative data to this study. My position as a volunteer also aided the qualitative research in this study by allowing trusting connections to be made with food parcel recipients prior to interviews. This trust was facilitated by the fact that all research participants had spoken to me at least once before, in a volunteer capacity; they already held trust and gratitude for Foodhall as an organisation; and many also had good relationships with other volunteers. This foundation of trust then facilitated the formation of reciprocal relationships, critical for community-based and ethical research (Maiter *et al.*, 2008). Although there was no financial incentive for participating in the interviews, I aimed to provide an open space for participants to speak about their experiences. The interviews followed a flow of conversation, and many participants appreciated this social interaction and open space to discuss their struggles during lockdown. However, it should also be noted that interviews happened over the phone, which perhaps created an additional barrier to connection.

Outside of my positionality as a Foodhall volunteer, I am a young woman from the U.S., which became clear over the phone. This often initiated 'small talk' about where I was from, which also in some cases led to conversations about missing family and more personal matters, as well as conversations about internationality and travel issues during COVID. This positionality in particular helped me connect to one American participant that was in the study.

From a sociological perspective, my positionality as an American, middle-class white woman also framed the understanding of the pervasion of neoliberalism in this work – which some say is the crux of the 'American mindset.' From childhood through my initial university education, this hyper-capitalist worldview focused on freedom, the individual, 'pulling yourself up by your bootstraps', providing for oneself, not paying for 'free-loaders' and the hunt for the 'American dream' pervaded my daily life. In some ways this subjectivity might be seen as withdrawing, but I believe it sets the foundation for my understanding of the deep feelings of stigma and shame that surrounded many participants.

Finally, my background as an engineer and natural scientist may seem to again be a hinderance to this research, but if anything, I believe that doing this research has allowed me to extend my subjectivity and scope as a food and agriculture researcher by drawing on the direct experiences of the people I aim to serve with my research. I believe my science-based background also provided me with more of a 'clean slate' when entering into the analysis of this research, which I see as a benefit. I think that this allowed for the outcomes and conclusions from this study to truly be grounded in the words of the participants, which was the aim, rather than by me imposing pre-existing theories and ideas onto the words of participants. Although several theories are explored in this research, these were investigated after drawing on the words of the participants in these interviews.

2.3.3 Quantitative data collection

For this study, quantitative data was collected from Foodhall volunteers about the project's emergency food response during March through July 2020. Volunteers consistently recorded data about the following during this time: number of delivered food parcels; postcodes being delivered to; number of people and households fed through the front-of-house service; number of adults, children, and households fed through the delivery service; number of cooked meals prepared; and number of volunteer hours. This data was provided to the author for analysis, with consent for reproduction and sharing; the data was also openly shared with all Foodhall volunteers and the National Food Service network.

2.3.4 Qualitative data collection

While providing important information, quantitative datasets do not provide information on the particular experience of the individual, often erasing personal experience behind a number or statistic. Thus, a qualitative methodology was also employed, with the aim to further understand the multiple dimensions and barriers associated with accessing food during the first months of the COVID-19 pandemic in England. By capturing the people's narratives, it is possible to uncover and develop the complexities and inequalities each individual faced, which are erased by statistical representations or simplified by theoretical frameworks (Hall and Holmes, 2020).

As the aim was to understand the experience of food insecurity during the pandemic, as well as the barriers and specific life circumstances that led to food insecurity among the study participants, it was important to engage with the lived experience of each individual. This allowed the researcher to gain important insights, which extend beyond characterizations of the individual based solely on socio-demographic context.

2.3.4.1 Participants and recruitment

All participants included in this study were receiving home-delivered food parcels in Sheffield from Foodhall during the first COVID-19 lockdown in the UK (end of March – July 2020). Interviews and recruitment took place over the phone in early August 2020, in compliance with social distancing guidelines. Participants were recruited by the author, who is both a researcher at the University of Sheffield and volunteer at Foodhall. Through her role as a volunteer, one task was to call food parcel recipients to notify them of updates to the food delivery service that was happening in early August 2020. During these calls, the opportunity was taken to mention the research project and recipients who expressed interest were called back later for an interview at the time they had requested. Criteria for participating was fluency in English and an age of eighteen or older; otherwise, participants were selected randomly from the food parcel recipients. Out of twenty-seven calls made to frequent food parcel recipients, seventeen expressed initial interest and fourteen ultimately completed the interview.

Table 61 provides a summary of the participants included in this study. The mean age of the participants was 50 (range 32-71). Half of the participants identified their sex as female, and the other half as male. Household size during the first UK COVID-19 lockdown ranged from one to thirteen people, with half of participants living alone. All participants relied on some form of fixed income during lockdown. Only one participant was employed before lockdown, although ineligible for furlough (government-subsidised payments from employers). Eight participants were not using food support before the pandemic, and out of these, six had never had to access food support at any time previously in their lives. Five participants reported a specific mental illness that became more challenging to cope with during lockdown, and many also reported high levels of stress and social isolation. Most described traumatic life events that contributed to these feelings, such as being in abusive relationships, being violently attacked, or experiencing the death or loss of a loved one. Twelve mentioned a specific physical illness or health challenge (besides COVID) that they were experiencing at the time of lockdown. Thus, all participants fell into a group that was considered vulnerable to food insecurity during COVID (Loopstra, 2020; The Food Foundation, 2021) – either by being unemployed, living with children, having a physical or mental illness, or being extremely clinically vulnerable to the virus; many fell within multiple of these categories.

Table 61 *– Summary of interview participants receiving food support from Foodhall*

Participant	Age range	Gender	Nationality	Employment status before lockdown	Income source during lockdown [a]	Shielding Status [b]	No. people in household during lockdown	No. children in household during lockdown	Used food support just prior to COVID? [c]	Struggled financially before or during COVID? [d]
William	60-70	Male	British	Retired	Pension	Self-Isolating	1	0	Yes	Neither
Randy	50-60	Male	British	Unemployed	Benefits	Shielding	3	1	No	During
Diane	40-50	Female	British	Unemployed	Universal Credit	Shielding	5	0	Yes	Both
Linda	40-50	Female	American	Employed (contract work)	Social Security (USA)	Following Guidelines	1	0	No	During
Caroline	70-80	Female	British	Retired	Pension	Shielding	3	0	No	During
Kathy	50-60	Female	British	Unemployed	Employment & Support Allowance	Self-Isolating	1	0	No	During
Liam	40-50	Male	British	Unemployed	Universal Credit	Self-Isolating	1	0	No	During
Gerald	30-40	Male	British	Unemployed	Universal Credit	Following Guidelines	1	0	Yes	Both
Paul	60-70	Male	British	Retired	Pension	Shielding	2	0	No	Neither
Martin	50-60	Male	British	Unemployed	Jobseeker's Allowance	Shielding	1	0	No	Both
Lilly	50-60	Female	British	Unemployed	Employment & Support Allowance	Following Guidelines	2	0	Yes	Both
Tara	30-40	Female	British	Unemployed	Benefits	Shielding	12	11	No	Unclear
Michael	40-50	Male	British	Unemployed	Benefits	Shielding	2	0	Yes	Both
Emily	30-40	Female	British	Unemployed	Benefits	Following Guidelines	1	0	Yes	Both

[a] In this column, "benefits" refer to state support payments, provided by the national government, where participants did not specify the support scheme used. When participants provided information on the specific kind of state support received (e.g., Universal credit, Employment and support allowance, or Jobseeker's allowance), this was specified.

[b] In this column, "shielding" refers to individuals advised by the UK Government to not leave their homes during lockdown, as they were classed as clinically vulnerable to the virus; "self-isolating" refers to individuals who themselves chose not to leave their homes; and "following guidelines" refers to individuals who followed Government lockdown guidance about isolation and social distancing, leaving their homes for limited reasons such as shopping (https://www.gov.uk/government/publications/full-guidance-on-staying-at-home-and-away-from-others).

[c] This column specifies if the individual was using a food support service just prior to the start of the pandemic.

[d] This column specifies if the individual was struggling financially just prior to COVID, only during the first few months of the COVID-19 lockdown (March-July 2020), if the individual was struggling before as well as during the COVID lockdown (designated as "both"), or if the individual was not struggling financially before or during the COVID lockdown (designated as "neither").

2.3.4.2 Interviews

Interview data collection within qualitative research involves a continuum from structured to unstructured. The strategy used for this data collection lies somewhere in the middle of this continuum, similar to a semi-structured or open conversation format. While a list of questions was used to guide the conversations, participants were also allowed and encouraged to speak freely about other topics related to their experiences during the first COVID-19 lockdown (Luo and Wildemuth, 2017). This gave the space for participants to bring forward experiences and issues that they felt were important. The use of this semi-structured interview approach is therefore particularly important to bring the experiences of those whose lives were affected by food insecurity to the forefront (Mapp, 2008).

Interviews were conducted individually over the phone with each participant during August 2020. This was following the relaxing of lockdown restrictions in July 2020, when pubs and restaurants were allowed to re-open and during the UK government's 'Eat Out to Help Out' scheme, which encouraged consumer support of local businesses. The lead author conducted the interviews as a volunteer who was knowledgeable about the Foodhall Project and who had already spoken to a variety of food parcel recipients over the project's helpline service.

On average, each interview lasted 44 minutes (range 14-138). Most questions were open-ended, in which participants were asked to discuss their experiences and methods of accessing food before and during the pandemic; experiences interacting with Foodhall and other similar food support projects; coping strategies to food insecurity; changes in financial security; and existence of social support networks and general social interaction with others during lockdown. Closed-ended questions were also used to gather sociodemographic characteristics, as well as information concerning levels of food insecurity, social isolation and loneliness, and community belonging and networks. These were respectively based on the U.S. Department of Agriculture's Six-Item Short Form Household Food Security Survey Module (USDA ERS, 2012); the Three-Item Loneliness Scale (Hughes *et al.*, 2004); and the UK Community Life Survey commissioned by the Office for Civil Society (UK Office for Civil Society, 2017). The interview guide used for this research is provided in Appendix I; however, it should be noted that this is simply a guide, as the interview was intended to feel conversational and other questions may have been asked to gather details or allow participants to expand about their experiences.

All interviews were audio-recorded and then transcribed. The interviewer took handwritten notes of the participants' responses during the interviews and recorded personal observations and reflections after the interviews. Pseudonyms were assigned to each participant to maintain confidentiality. Informed verbal consent was recorded for each participant before the interviews. The study was reviewed and approved by the University of Sheffield Research Ethics Committee.

2.3.4.3 Analysis

Data was coded and analysed in two stages, using a combination of qualitative description and grounded theory approaches. This allowed for personal experience to be examined and then used to inform ideological context.

Transcripts were analysed using NVivo 12.0 Pro, a computer software for qualitative data management, which allowed for data to be organized by codes and major themes (QSR International Pty Ltd., 2018). The initial analysis was conducted by the interviewer, using descriptive categorization to identify the words and phrases used by participants which were of research interest (Sandelowski, 2000; Moghaddam, 2006). This low-inference interpretation of the data involved examining, summarizing, categorizing, and grouping the data into descriptive items of meaning, in relation to the broad themes of the interview questions. These themes included: accessing food, food insecurity, diet and cooking, interactions with the Foodhall Project and other food services, social networks and interactions, mental health, physical health, and financial security. This initial descriptive exercise allowed for the researcher to connect with and familiarize them self with the data. It also provided an organized data set for subsequent analysis using a grounded theory approach.

The second phase of the analysis involved identifying and naming concepts to reveal the meanings within the descriptive categories, a stage referred to as axel coding (Moghaddam, 2006). The data was hierarchically organised into overarching codes to develop theoretical relationships between categories and subcategories (Strauss and Corbin, 1998). This stage was based upon a grounded theory methodology, in which, theoretical constructs are inductively derived from the qualitative analysis of data (Corbin and Strauss, 2012). This approach was selected since the aim of the research was to capture participants' lived experiences, and thus it was important that the interpretation of the data was grounded in the words and experiences of participants as they reported them. This is in line with previous studies which aim to capture the lived experiences of participants (Runnels, Kristjansson and Calhoun, 2011). To facilitate this, two researchers (including the author and the academic supervisor on this project) reviewed the descriptive codes to extract broader relationships that provided a theoretical understanding of each phenomenon's occurrence. These themes were then reviewed and discussed between the researchers.

2.3.4.4 Knowledge exchange

A key part of this research project was sharing the results in a timely manner, in order to inform community food responses, local city initiatives, and also national government measures during the pandemic. It was important to share results as quickly as possible, so these groups could understand the barriers people were facing to accessing food and use this to inform future decision-making. Therefore, the author allocated time to preparing reports, presentations, and responding to queries to inform the coronavirus crisis response.

Following the completion of the interviews and all data analysis, the author discussed the main findings with Foodhall staff and volunteers, providing key recommendations to inform the project's future operations. The author produced a summary report targeted toward Foodhall and other community food projects, with the aim to share the key findings and recommendations that could inform the operation and decision-making process of these projects in the future (Kennard, 2023). This report was shared with the National Food Service network, which comprises a range of community food projects across the UK. Results were also shared with the Sheffield City Council's Overview and Management Scrutiny Committee, to inform their review of food poverty across Sheffield (February-March 2021) (Sheffield City Council, 2021). Written evidence was additionally provided to the

Environment, Food and Rural Affairs (EFRA) parliamentary committee, for their inquiry on the Covid-19 pandemic and food supply (EFRA, 2020b). Written evidence for the inquiry was prepared chiefly by the author, along with other Foodhall volunteers, on behalf of Foodhall. After the initial response to the 2020 inquiry (https://committees.parliament.uk/writtenevidence/3319/html/), EFRA requested further information by email for their 2021 follow-up to the initial inquiry (EFRA, 2021). This was then provided by the author (https://committees.parliament.uk/writtenevidence/21848/html/).

The author further contributed to knowledge exchange at Foodhall following the completion of this research study. In realising the importance of gathering stories and data during the COVID lockdown, Foodhall staff requested further support from the author to put systems in place to consistently collect data and track impact of the project. The author was financially supported to do this in the fall of 2021 through the University of Sheffield's Postgraduate Researcher Experience Programme grant.

Chapter 3: Evaluating environmental impacts from local agriculture using lifecycle assessment

This Chapter presents results of the lifecycle assessments performed for tomato and kale production on different scales of local farms, employing organic and conventional management practices, in two case study locations: Georgia, USA and England, UK. Details of the methodology and the lifecycle inventories for farms are provided in Section 2.1. This study aims to provide insight to the influence of farm scale, distance to consumer, and management practices on the environmental impacts generated from crop production, processing, and transport. The agricultural models evaluated include: urban organic; peri-urban organic; rural organic; and rural conventional farms.

3.1 Results: Lifecycle impact assessment

Through lifecycle impact assessment (LCIA), the lifecycle inventories presented in Section 2.1.7 are translated into environmental impacts. The ReCiPe 2016 v.1.05 method is used for this study, which provides results for a total of eighteen midpoint impact categories. Seven impact categories are discussed at length within this Results section, which include: global warming potential (kg CO_2 eq), fine particulate matter (PM) formation (kg PM2.5 eq), terrestrial acidification (kg SO_2 eq), freshwater eutrophication (kg P eq), marine eutrophication (kg N eq), water consumption (m^3), and land use (m^2a crop eq). The main substances that contribute to impacts for these categories, throughout all crop lifecycles assessed in this study, are presented in Table 62.

Table 62 *– Main contributing substance for each impact category*

Impact categories	Unit	Main contributing substances [a]
Global warming	kg CO_2 eq	N_2O, CO_2, CH_4 (various forms), SF_6, C_2H_6 (various forms) (all to air)
Fine particulate matter formation	kg PM2.5 eq	NH_3, PM < 2.5 μm, SO_2, NO_x, NO_2, SO_x, SO_3, NO_3^- (to air), NO (all to air)
Terrestrial acidification	kg SO_2 eq	NH_3, SO_2, NO_x, NO_2, SO_x, SO_3, NO_3^-, H_2SO_4, NO (all to air)
Freshwater eutrophication	kg P eq	P (to water), PO_4^{3-} (to water), P (to soil)
Marine eutrophication	kg N eq	NO_3^-, NH_4^+, N, NO_2^-, N, NO_2 (all to water)
Water consumption	m^3	All water withdrawals from ground or surface water, not including rain
Land use	m^2a crop eq	All land occupations and transformations

[a] Contributing substances are listed in descending order as seen on most farms, although exact order varies for each crop lifecycle.

These seven impact categories have been selected for in-depth discussion because of their timely relevance to current pressing environmental issues, such as climate change and biodiversity loss, and also because they are known to be particularly impacted by the food system. Particularly, global warming potential provides insight to food system impacts on climate change, which in turn threatens future food production, as well as human and environmental health (Vermeulen, Campbell and Ingram, 2012; FAO, 2017; Crippa *et al.*, 2021). Fine particulate matter formation provides insight to agricultural contributions to pollution that can affect human health and livelihood (Pozzer *et al.*, 2017; Wyer *et al.*, 2022). Terrestrial acidification, freshwater eutrophication, and marine eutrophication together highlight types of environmental pollution that are commonly spurred by fertiliser use in agriculture and can lead to species and biodiversity loss, as well as impacts on human health (e.g., through water pollution and increased disease) (Azevedo *et al.*, 2013; Hwang, 2020; Xu *et al.*, 2020). In particular, the acidification of soil, along with the loss of plant and soil life that comes with it, can in turn negatively impact agricultural productivity (Zhang *et al.*, 2016; Xu *et al.*, 2020). Finally, water consumption and land use provide insight to the use of critical and finite resources within the food system, for which agricultural production is known to be a major driver (Pfister *et al.*, 2011). Together, these impact categories present a holistic view of some of the key factors that both impact and are impacted by the food system, allowing for a systems-level insight to the links between food system sustainability and human and ecosystem health. Although these impact categories will be the main focus, the results from the other midpoint impact categories are also included within Appendix B.

These seven impact categories are assessed in terms of the functional unit of the LCA, which, as described in Section 2.1.1.1, is one kilogram of kale or tomato transported to the final point of sale. This results section includes environmental impact results from the lifecycles of two different crops (kale and tomatoes) from farms in two different countries (Georgia, USA and England, UK). Farms are categorised into the following groups: urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). Results are first presented in terms of farm category groups (Figure 10-Figure 23), and then impacts from individual farms and the specific processes and practices that contribute to these impacts are presented afterwards (Figure 24-Figure 51). The last results section includes the outcomes from additional sensitivity analyses, as described in Section 2.1.6.

For the first two sets of results, impacts are presented for all three allocation scenarios (Scenarios 1-3), as described in Section 2.1.5. These scenarios differ in the allocation of compost production burdens and of combined heat and power, or CHP (relevant to the UK conventional farm only). Briefly, Scenario 1 is considered the default and allocates the burdens of the composting process to farms that use compost as a fertiliser input. In Scenario 2, the burdens of the composting process are allocated to the lifecycle generating green waste or crop waste; thus, additional burdens are seen on farms that compost their crop waste or green waste, and composting burdens for compost inputs are not included. For both of these scenarios, the surplus electricity generated by CHP is allocated based on energy allocation. Finally, Scenario 3 accounts for avoided burdens of compost and CHP using the substitution method. Avoided burdens from the municipal waste stream are subtracted from composting burdens for compost that is used as a fertiliser input, and the avoided burdens from generating electricity via the GB national grid are subtracted from the burdens of CHP.

3.1.1 Results for farm categories

Impact assessment results from individual farms are first presented as groups based on farm categories for each country (U.S. and UK) and crop lifecycle (kale and tomatoes). This allows for the evaluation of trends based on farm models and scales of locality. Results have been presented as box plots, displaying the farm category mean, median, and interquartile range (25-75%); individual farm data is also represented as singular points within each group. These plots thus allow for variation between and within farm categories to easily be examined. In cases where only one data point (one farm) exists within a certain category, results will be depicted as a single line and dot (as done for the medians and means within the box plot).

These box plots are provided for each of the seven main impact categories. Six plots are provided per page for a specific impact category and crop lifecycle. This includes three allocation scenarios set side-by-side, with results for U.S. farms at the top of the page and UK farms at the bottom. Figure 10 through Figure 16 provide results for the kale lifecycles on U.S. and UK farms, for the impact categories of global warming, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, water consumption, and land use, per kg of kale transported to the final point of sale. Similarly, Figure 17 through Figure 23 provide an overview of the same impact categories for tomato lifecycles on U.S. and UK farms, per kg of tomato transported to the final point of sale. In Appendix B, Table 75 and Table 76 provide results for all eighteen midpoint categories as generated from the ReCiPe 2016 method, for U.S. and UK kale production respectively, as averaged per farm category and per scenario. Similarly, Table 77 and Table 78 in Appendix B provide the same results for U.S. and UK tomato production, respectively.

3.1.1.1 Comparisons of farm categories & countries: Scenario 1

Scenario 1 is considered the default scenario used throughout this LCA, so this scenario will first be used to examine differences in impact data between farm categories and countries within this section. Section 3.1.1.2 will then go on to discuss differences observed between the scenarios and how this influences the results.

3.1.1.1.1 Kale

For U.S. kale production in Scenario 1, the rural conventional farm category (n=1) generally has the lowest impacts compared to the urban (n=1), peri-urban (n=4), and rural (n=3) organic farm categories. This is seen for fifteen out of the eighteen environmental impact categories, including global warming (Figure 10), fine particulate matter formation (Figure 11), terrestrial acidification (Figure 12), freshwater eutrophication (Figure 13), water consumption (Figure 15), and land use (Figure 16), among others (Table 75). However, for marine eutrophication (Figure 14), the peri-urban organic and rural organic farm categories both have lower average impacts in comparison to the conventional. Overall, the urban organic farm category (n=1) tends to have the highest impacts across impact categories, seen in twelve out of the eighteen categories.

The magnitudes of difference between conventional and organic categories vary, but are especially profound for freshwater eutrophication, global warming, and fine particulate matter formation. For freshwater eutrophication, the organic farm categories display impacts anywhere from 4-10x the impact levels of the conventional farm, with the greatest difference seen when comparing to the urban organic farm. For the global warming and fine particulate matter formation categories, the conventional farm has impacts that are at least half of rural organic average, one-third of peri-urban organic average, and one-fifth of the urban organic farm. It is thus seen that the urban farm displays exceptionally high impacts in many cases.

When comparing the urban farm to the other organic farm category averages for global warming, fine particulate matter formation, and freshwater eutrophication, the urban farm's impacts are approximately 1.5-2x higher than the other categories' averages. Even more profound differences are seen in the terrestrial acidification category, where the urban farm impact level is 3x higher than peri-urban average and 5x higher than the rural organic average, as well as for marine eutrophication, where the urban impact level is 16x higher than the peri-urban average and 7x higher than the rural organic average.

Considering farm category averages, a basic trend of decreasing impacts with increasing farm scale and rurality is observed for several impact categories. This is seen in how the urban organic farm tends to have the highest impacts, peri-urban organic the second highest, rural organic the third highest, and rural conventional the lowest in impact categories such as global warming (Figure 10), fine particulate matter formation (Figure 11), and terrestrial acidification (Figure 12), as well as several other toxicity categories (Table 75). However, it should also be noted that the peri-urban organic farm category includes one farm that sits as an outlier, with impact levels often similar to or even higher than the urban organic farm. Thus, the peri-urban farm category average is driven higher by this outlier, and if this farm was excluded, the category average would be much lower and similar to the rural organic category average for global warming, fine particulate matter formation, and terrestrial acidification, therefore somewhat diluting the trend of decreasing average impacts with increasing scale and rurality by making the peri-urban organic farm model more environmentally attractive. In some cases, the peri-urban organic farm category average (including all farms) is already lower than the rural organic average, such as for freshwater and marine eutrophication (Figure 13 and Figure 14, respectively). Finally, for water consumption (Figure 15) and land use (Figure 16), both the urban and peri-urban organic farm categories perform better than the rural organic farms, with the rural conventional farm still having the lowest impacts.

For UK kale production, the rural organic farm category (n=2) on average displays the lowest impacts across fourteen out of eighteen impact categories (Table 76), as compared to the urban organic (n=1), peri-urban organic (n=4), and rural conventional (n=5) farm categories. This is seen for impacts including global warming (Figure 10), fine particulate matter formation (Figure 11), freshwater eutrophication (Figure 13), and marine eutrophication (Figure 14), among others; however for terrestrial acidification, land use, and water use, the conventional farm category impacts are lower. The magnitude of differences between the best performing farm category (rural organic) and the other farm categories are much less profound than as seen between the U.S. farms, indicating lower variation in impacts between farm categories in UK kale production versus U.S. For example, the rural organic farm category average for global warming potential is approximately one half to one-third that of all other farm categories, for which impacts are fairly similar; this is in contrast to U.S. kale production, where the organic farm categories' global warming potentials are up to 5x higher than the rural conventional average. Similarly, for freshwater eutrophication, the UK rural organic average is only about 1.3-2.5x lower than the values of other categories, whilst in the U.S., the rural conventional average is at minimum 4x lower and at maximum 10x lower than other farm categories.

The overall better environmental performance of the rural organic farm category in UK kale production is also in obvious contrast to the case in the U.S., which saw conventional production as generally having the lowest impacts. However, the low impacts achieved by UK rural organic production in most categories is set against relatively higher impacts seen in a few other categories including terrestrial acidification (Figure 12), water consumption

(Figure 15), and land use (Figure 16). This is similar to results from U.S. kale farms, where the rural organic categories also had relatively high impact levels for land use and water consumption compared to the conventional and other organic farm categories.

Although the rural organic farm did consistently have the lowest impacts for most categories in the UK, there is high variation in performance between the other three farm categories depending on the impact in question. Thus, the general trend of decreasing impacts with increasing scale and rurality that was observed for U.S. kale production is less obvious in the UK. Urban and peri-urban farm category averages are similar for global warming, fine particulate matter formation, and water consumption. For freshwater eutrophication and terrestrial acidification, the peri-urban average is markedly higher than the other organic categories, whilst the urban farm category has the highest impacts for land use and marine eutrophication. Regarding conventional production, this farm category actually has the lowest impacts overall for the terrestrial acidification and land use, although has the highest impacts for other categories such as the environmental ecotoxicities (Appendix B, Table 76). This points to significant trade-offs between environmental impacts for all farm categories, although significantly less so for rural organic.

Comparing impacts from kale production between the U.S. and UK, it is seen that U.S. production generally results in higher impacts when comparing the same farm categories across countries, with differences being more pronounced among organic farm categories rather than the conventional. For example, average global warming impacts (kg CO_2 eq per kg sellable kale) for U.S. versus UK production were, respectively: 5.9 vs. 1.6 for urban farms; 3.9 vs. 1.7 for peri-urban farms; 2.8 vs. 0.60 for rural organic farms; and 1.1 vs. 1.4 for rural conventional farms (Figure 10). Exceptions to this are seen particularly in the land use category (Figure 16), where U.S. farms across the board actually have lower land use impacts. Large differences are especially seen between urban organic farms, where the UK urban farm has 8x higher land use than the urban farm in the U.S., and conventional farms, where UK farms have 3.8x higher land use than in the U.S.

Viewing the U.S. and UK results together, it can be seen that generally, high variation is observed within and between organic farm categories. This variation is especially seen within the peri-urban organic farm category, although this farm category also has the highest number of farms compared to the other organic farm categories. Considering conventional production, for the U.S., there was only one farm in the conventional category, so variation cannot be observed; however, in the UK, the five conventional kale farms generally had lower variation compared to the organic farms when viewed all together. This points to the wider range of practices employed by organic farms on different scales, in contrast to the more standardised and similar set of practices employed across conventional farms that likely generate similar impact outcomes. Higher variation among UK conventional farms is observed for some impact categories, however, such as fine particulate matter formation, eutrophication categories, mineral resource scarcity, and water consumption.

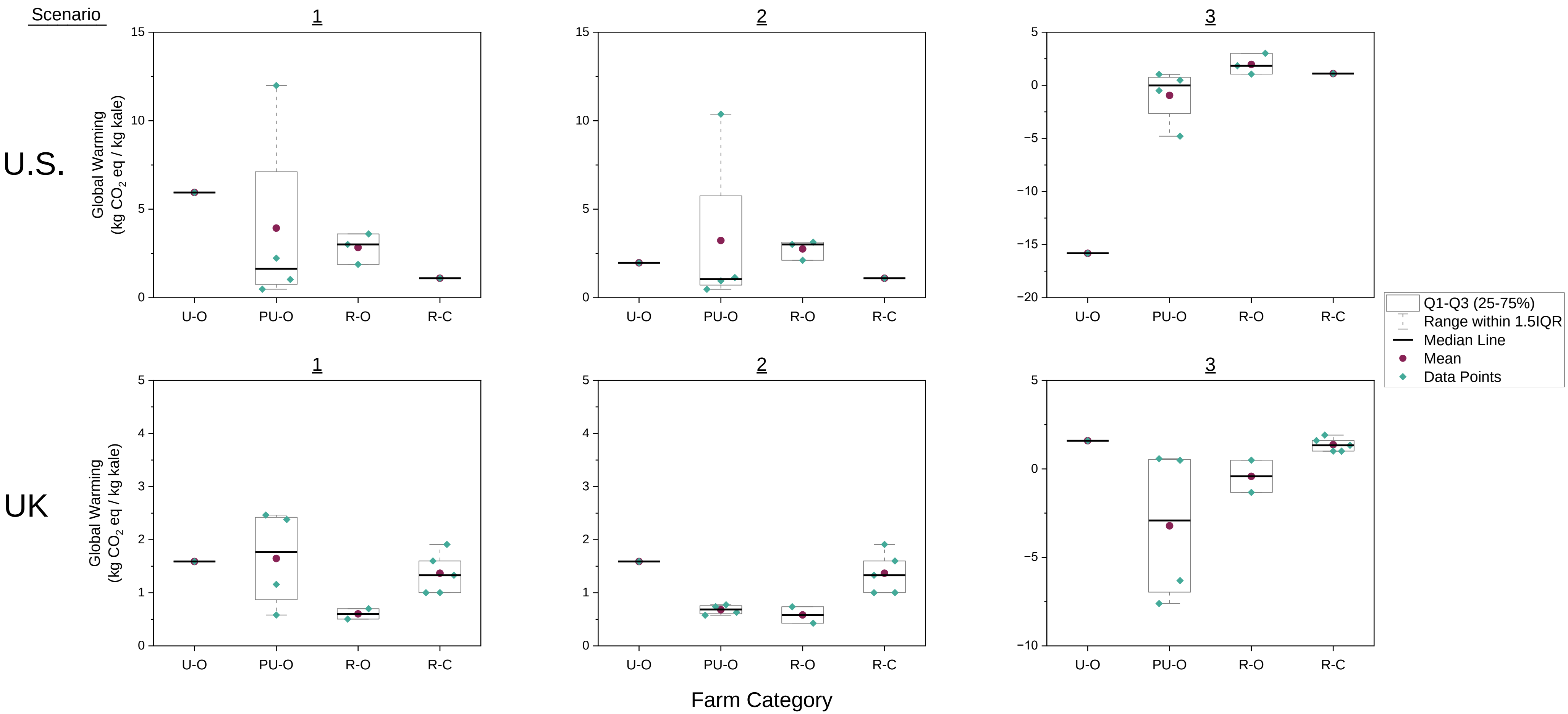

Figure 10 *– Global warming potential per kg kale transported to final point of sale (kg CO2 eq / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Note that axis scales differ between U.S. and UK plots and also for Scenario 3 plots, due to negative impacts.*

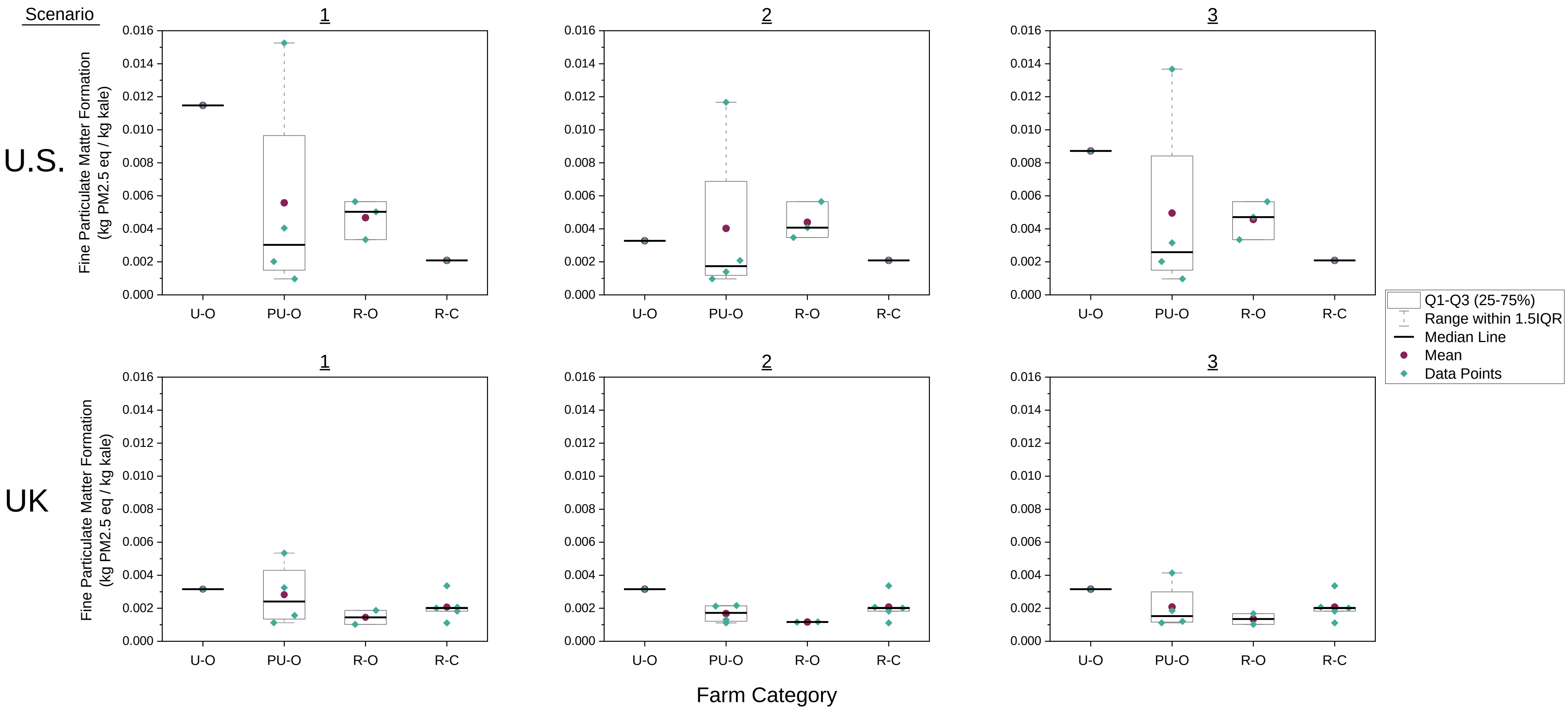

***Figure 11** – Fine particulate matter (PM) formation per kg kale transported to final point of sale (kg PM2.5 eq / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

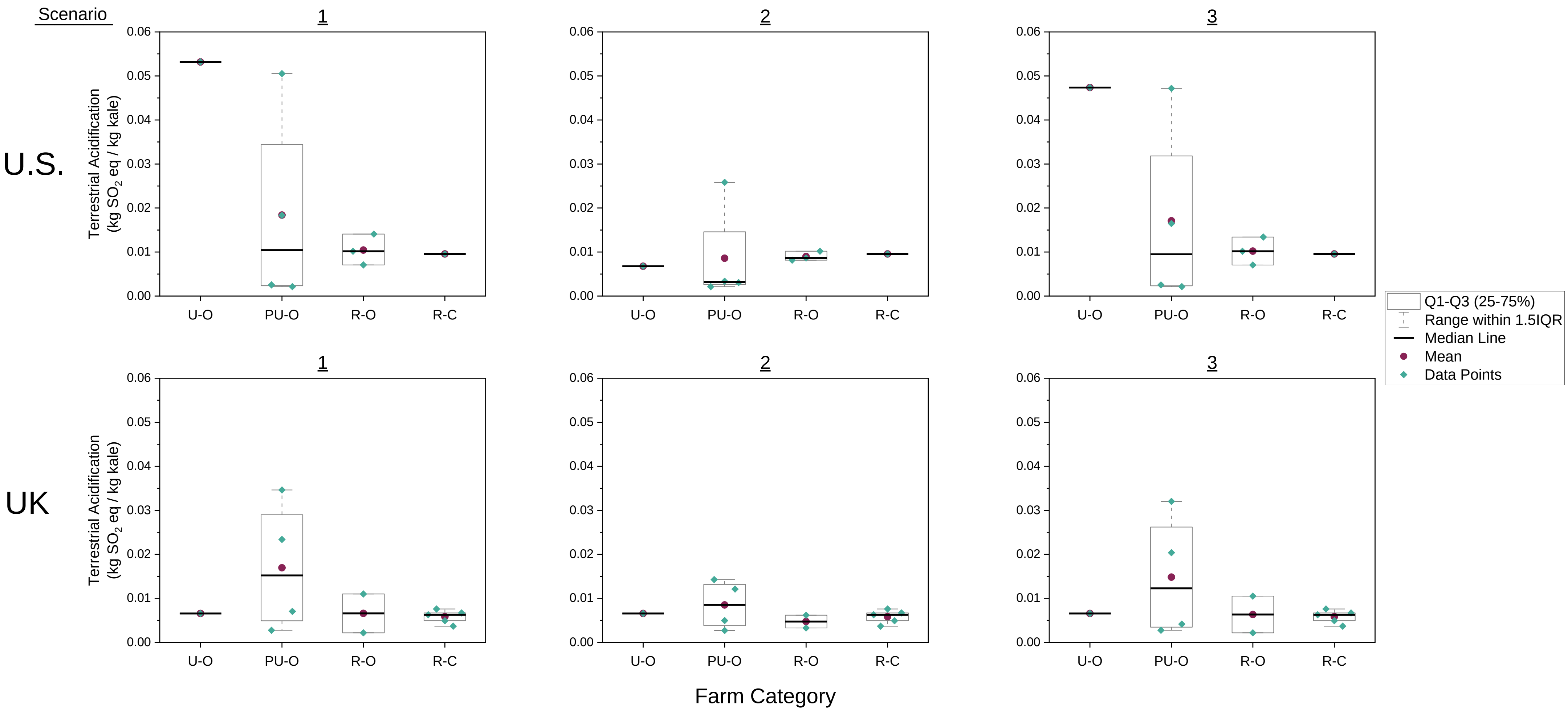

Figure 12 *– Terrestrial acidification per kg kale transported to final point of sale (kg SO_2 eq / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

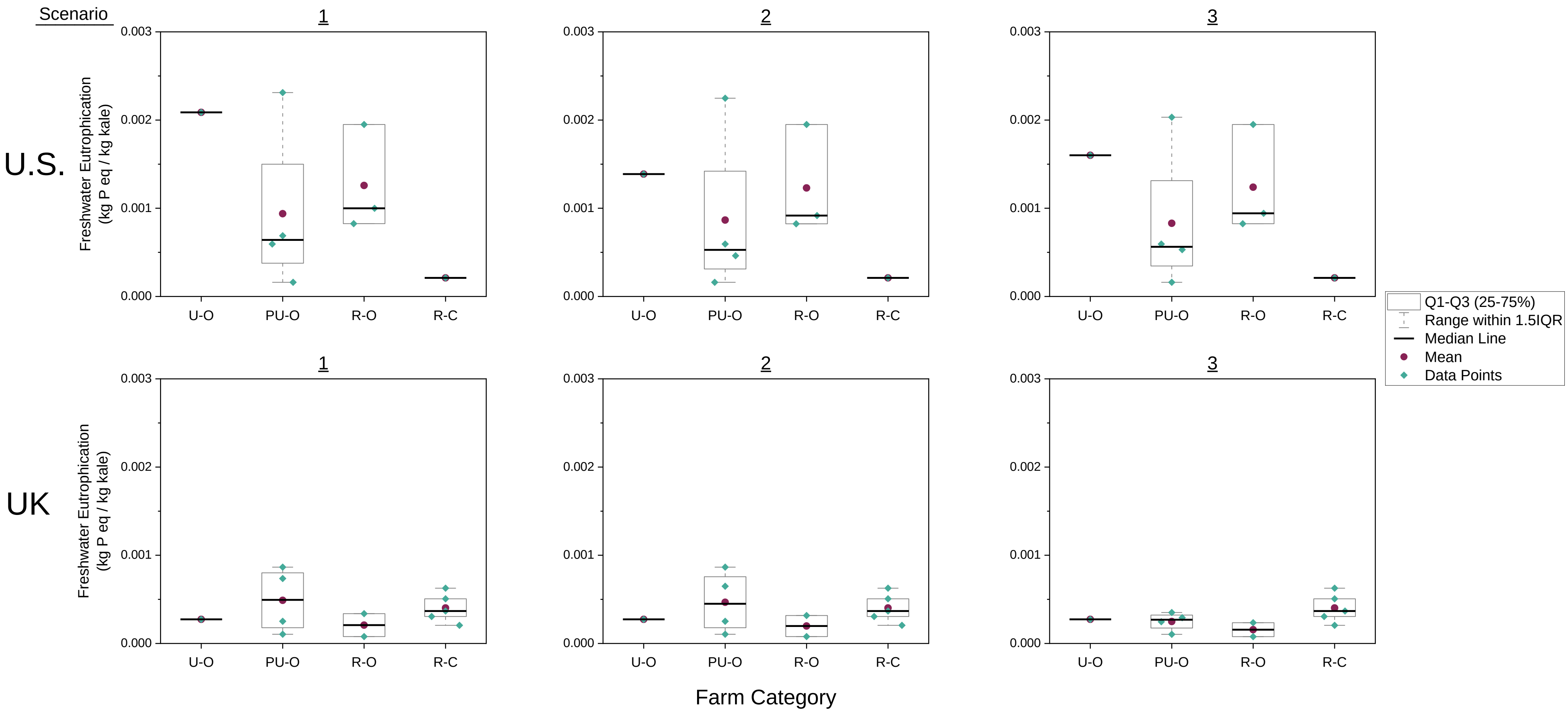

Figure 13 *– Freshwater eutrophication per kg kale transported to final point of sale (kg P eq / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

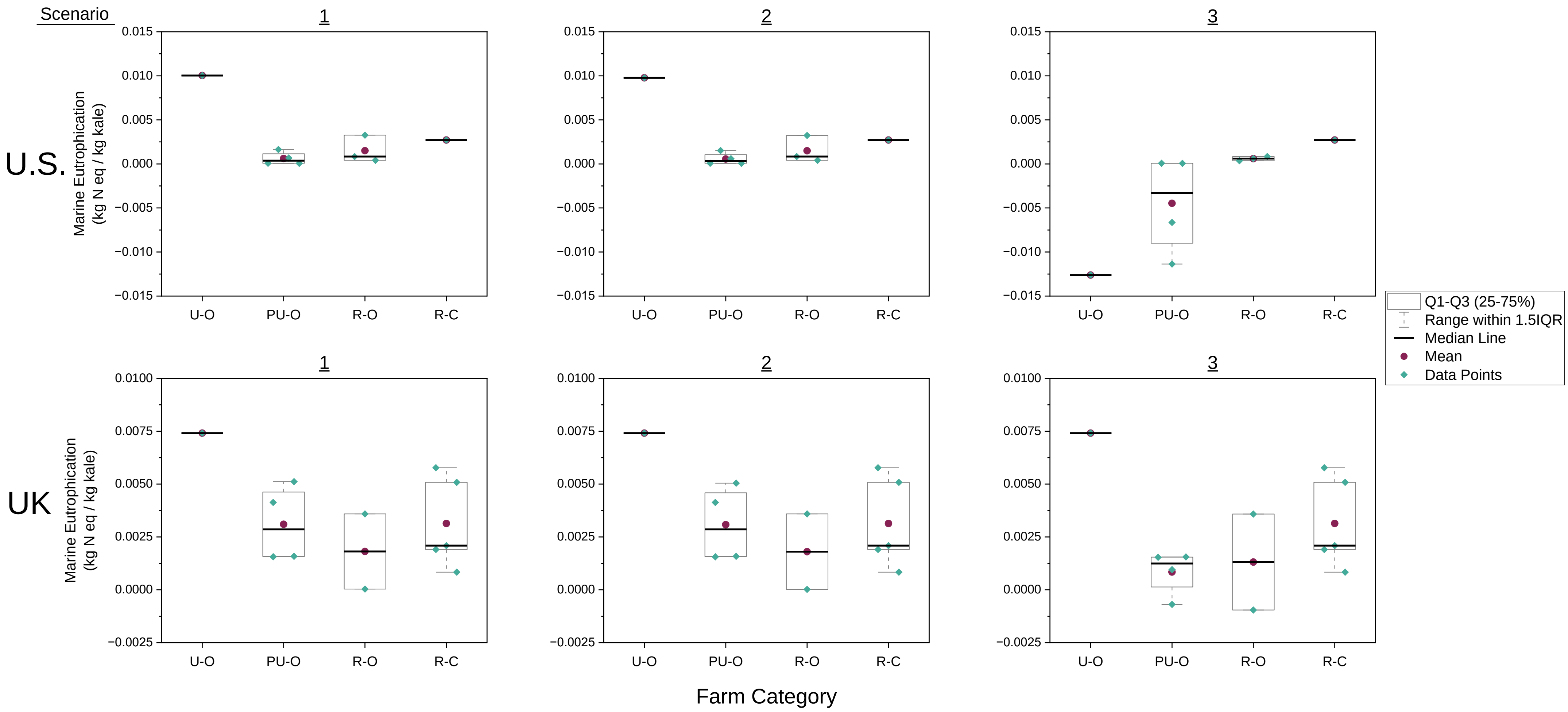

__Figure 14__ - Marine eutrophication per kg kale transported to final point of sale (kg N eq / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales differ between U.S. and UK plots.

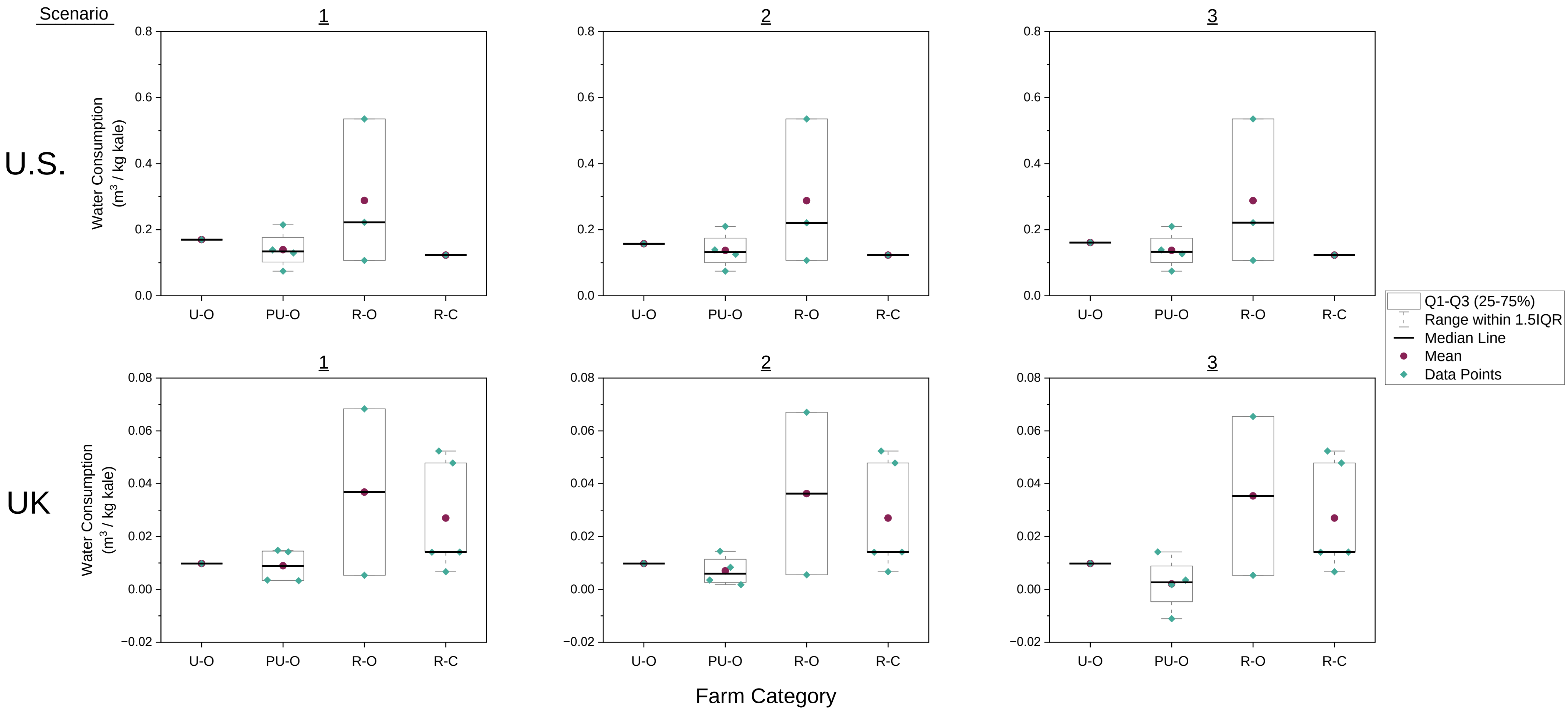

***Figure 15** – Water consumption per kg kale transported to final point of sale (m^3 / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales differ between U.S. and UK plots.*

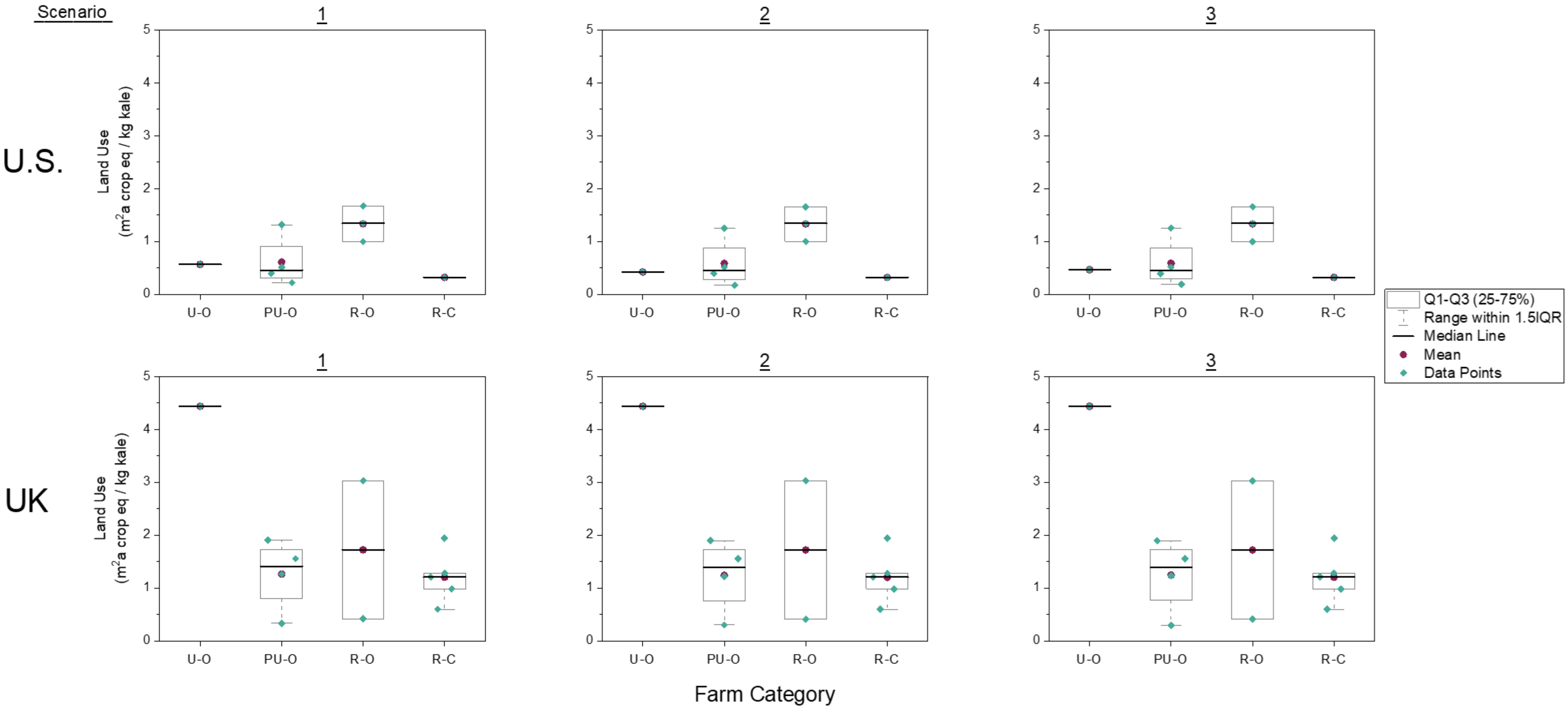

Figure 16 *– Land use per kg kale transported to final point of sale (kg m^2a crop eq / kg kale), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

3.1.1.1.2 Tomatoes

Similar to kale, for tomato production in the U.S., the rural conventional farm category (n=2) generally has the lowest average impacts compared to the urban (n=1), peri-urban (n=3), and rural (n=3) organic farm categories. This was seen across all impact categories with only one exception, where for water consumption the rural conventional farm average is slightly higher than the urban farm's water consumption (Figure 22). Large differences between the conventional and organic farm category averages are observed for global warming (Figure 17), fine particulate matter formation (Figure 18), and terrestrial acidification (Figure 19), where organic farm categories have impacts at least 4x and up to 12x higher. Large variation is also observed for the human toxicity categories, with organic farms having impacts approximately 10x higher for human non-carcinogenic toxicity and 15-30x higher for human carcinogenic toxicity, depending on the categories (Appendix B, Table 77).

The basic trend of decreasing impacts with increasing farm scale and rurality is not observed as clearly for U.S. tomato production as it was for kale production, and is only really seen in the case of marine eutrophication (Figure 21). Otherwise, the peri-urban farm category generally has the lowest impacts out of the organic farm categories, while the rural organic farm category often has the highest impacts on average for most impact categories (thirteen out of eighteen). This is in contrast to U.S. kale production, where the rural organic category generally performs better than the more urban organic farm categories. This difference is driven largely by the higher variation in growing practices employed by the rural organic farms for tomato production and the higher resource use, which translate to relatively higher variation and a higher farm category average for rural organic farms than seen for kale production. When comparing the rural organic average to the peri-urban organic average, differences are particularly seen for global warming, fine particulate matter formation, freshwater eutrophication, terrestrial acidification, and water consumption, where the rural organic farm category average is approximately double that of the peri-urban. For water consumption in particular, a clear trend of increasing impacts with increasing rurality is observed among the organic farms (Figure 22), with the rural conventional farm category having average impact levels similar to the urban and peri-urban organic farm categories.

In contrast to the rural organic farms generally having the highest impacts for U.S. tomato production, in the UK the rural organic farm category (n=2) generally shows the lowest average impacts in comparison to the urban organic (n=2) and peri-urban organic (n=4) farm categories, as well as the rural conventional cases using conventional heating and electricity (n=1) or CHP (n=1). This is observed for thirteen out of eighteen impact categories (Appendix B, Table 78), including global warming (Figure 17), fine particulate matter formation (Figure 18), terrestrial acidification (Figure 19), freshwater eutrophication (Figure 20), marine eutrophication (Figure 21), and water consumption (Figure 22). This is similar to trends observed for UK kale production, where the rural organic farm category also has the lowest overall impacts.

However, for UK tomato production the relationship between organic and conventional cultivation is more complex due to the drastically different growing practices between the two systems. Conventional production is characterised by hydroponic cultivation in heated glasshouses, whilst organic production is characterised by growing in soil within polytunnels. Thus, the differences in impact levels observed between organic and conventional production in the UK are quite high for certain categories. This is especially distinct for global warming (Figure 17), mineral and fossil resource scarcity, and several ecotoxicity categories (Appendix B, Table 78). The magnitude of differences between conventional production and the rural organic farm category are the most severe. For example, the global warming potentials of rural conventional production via CHP and conventional energy sources are 13 and 14x higher than the rural organic average, respectively, but only 2-3x that of the urban and peri-urban farm category averages.

There are also differences observed between the two cases of conventional production – that using natural gas for heating and the national grid for electricity (signified as R-C NG) and that using combined heat and power (R-C CHP). In all impact categories, the use of CHP results in lower impacts in comparison to using conventional energy sources. However, the magnitude of this difference varies across impact categories. This difference is more significant for impact categories such as fine particular matter formation and terrestrial acidification, where using CHP over conventional energy sources results in impact decreases of 17% and 24%, respectively. However, in other cases the differences are less obvious. For example, the use of CHP over conventional energy sources only results in an 11% decrease for global warming potential; 9% for land use; 6% for freshwater eutrophication; 2% for water consumption; and 1% for marine eutrophication.

The impacts from conventional production using traditional energy sources (R-C NG) is highest for eleven out of eighteen impact categories, but in many cases the urban farm category displays similarly high impacts. This is seen particularly for the eutrophication categories, where the urban farm category average and the two cases of conventional production have impacts approximately 5-6x higher than the rural organic average for freshwater eutrophication and 6-11x higher for marine eutrophication. Additionally, the urban farm category actually has the highest average impacts for six out of eighteen impact categories (Appendix B, Table 78). This is seen for impacts such as fine particulate matter formation (Figure 18), marine eutrophication (Figure 21), terrestrial acidification (Figure 19), and land use (Figure 23). For the latter two impact categories, the peri-urban farm category average is also higher than the conventional averages, but often with relatively high variation seen between individual peri-urban farms, making the differences less clear.

For land use in particular, the urban farm category has the highest overall impact whilst the conventional production has the lowest impacts, due to the intensive hydroponic production (Figure 23). Comparing between organic farms, it is also interesting to note that, unlike for all other crop lifecycles (U.S. kale and tomato production and UK kale production), the rural organic farm category had some of the lowest land and water use impacts out of the other organic farm categories. This is perhaps due to the fact that all UK organic farms grew tomatoes using a similar production method – trellised in polytunnels – and thus, the larger scale of rural organic farms potentially showed more efficient resource use, in comparison to varied types of polytunnel and field production as used for kale production and U.S. tomato production.

Comparing U.S. and UK tomato production across the same farm categories, it can be seen that generally, UK organic farm categories have lower impacts on average than their U.S. counterparts. For conventional production, the hydroponic production of tomatoes in the UK shows higher impacts than U.S. field-based tomato production for most impact categories (fourteen out of eighteen). For example, for global warming, UK conventional production has impacts of 4.19 and 3.73 kg CO_2 eq per kg sellable tomato for production with conventional energy sources and CHP, respectively, while U.S. conventional production has an average global warming potential of 0.35 kg CO_2 eq (kg tomato)$^{-1}$ (Figure 17). However, U.S. conventional production had approximately 1.5x higher land use (Figure 23) and 1.4x higher water use (Figure 22) than UK conventional production.

Further, it can actually be seen that in some cases UK hydroponic production outperforms U.S. organic production. For example, the average global warming potential for the U.S. rural organic farm category is essentially the same as UK hydroponic production, when using conventional energy sources. UK hydroponic production (for both energy types) also has lower impacts than U.S. organic production (for all farm categories) for fine particulate matter formation, terrestrial acidification, marine eutrophication, land use, and water use, although differences are generally less pronounced when comparing to peri-urban farms, which is the lowest impacting U.S. organic farm category for tomatoes.

When comparing U.S. rural conventional production (the best performing U.S. farm category) to UK rural organic production (the best performing UK farm category), in many cases the impact levels between the two are quite similar. However, U.S. conventional production has impacts 7x higher than that of UK rural organic for marine eutrophication and 3x higher for water consumption, although this is expected due to the much hotter climate in Georgia, USA compared to England. Overall, this suggests that sustainability differences between organic and conventional agriculture are not clear-cut, but depend on specific management practices and local context.

As similarly observed for kale production in the U.S. and UK, the high variability in impacts between organic farms (both within and between categories) is clearly observed for tomato production. For the U.S., the highest variation is observed within the rural organic farm category, whilst for the UK, both the urban and peri-urban farm categories display variation, although generally to a lesser extent than seen across U.S. farms. For U.S. conventional production, two farms were considered, which show generally similar results. This again points to the idea that conventional farms portray lower variation likely due to the use of a more standard set of practices, and indeed throughout the lifecycle inventories it can be confirmed that both U.S. conventional tomato farms used a similar set of growing practices, pesticide regimes, and trellising and irrigation setups. However, the differences between conventional production for UK farms cannot be discerned from this data, since this dataset is based on information from only one farm, although considering two energy sources. Still, if more farms had been considered, it can be assumed that there would be at least the amount of variation displayed between these two conventional cases, as some tomato glasshouse production in the UK use conventional energy sources and others use CHP. Overall, the high amount of variation between the organic farms in both the kale and tomato lifecycles suggests that clear conclusions on environmental sustainability cannot be drawn between farm models based simply on organic status or local scale. More consideration into individual farm practices is required.

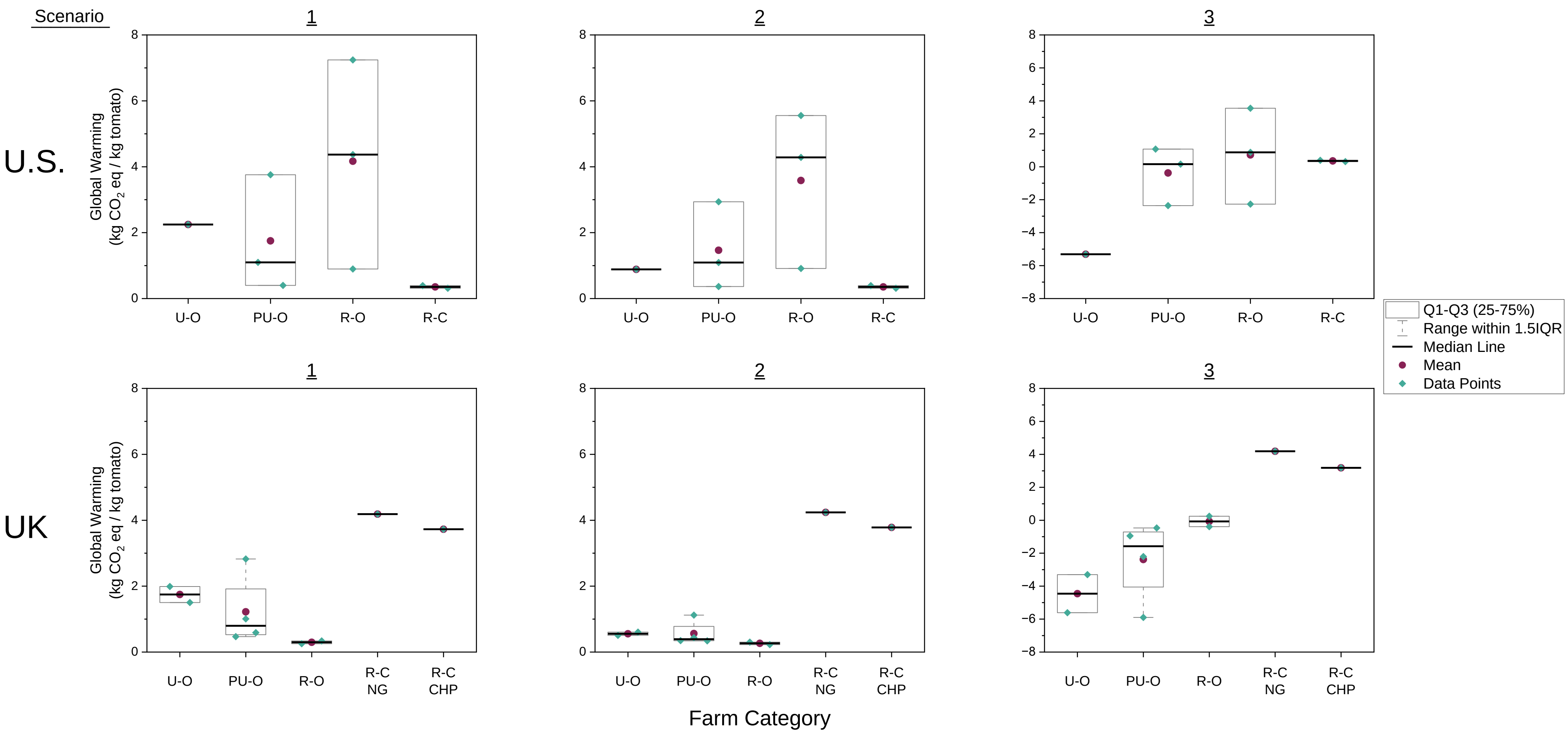

Figure 17 *- Global warming potential per kg tomato transported to final point of sale (kg CO_2 eq / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Note that axis scales differ for Scenario 3 plots, due to negative impacts.*

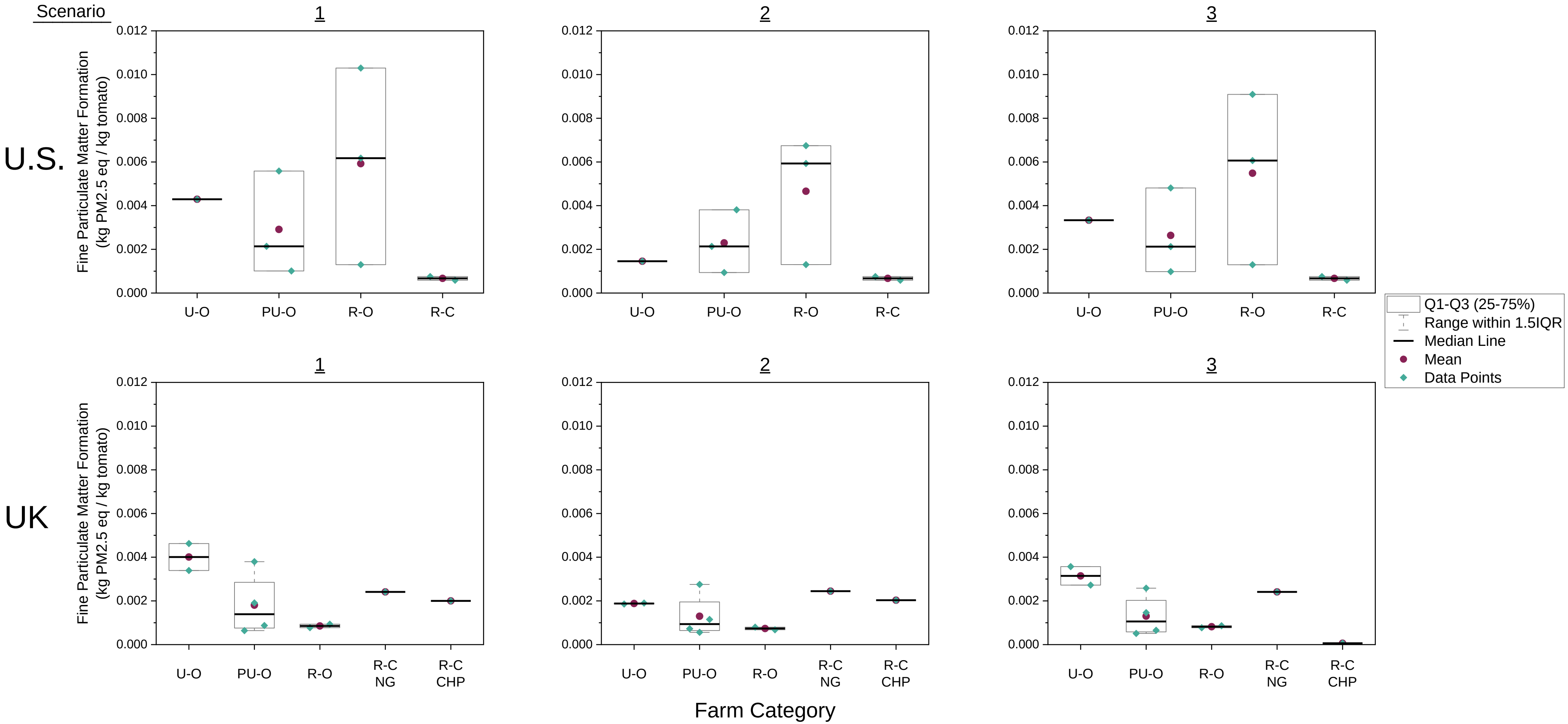

Figure 18 *– Fine particulate matter formation per kg tomato transported to final point of sale (kg PM2.5 eq / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

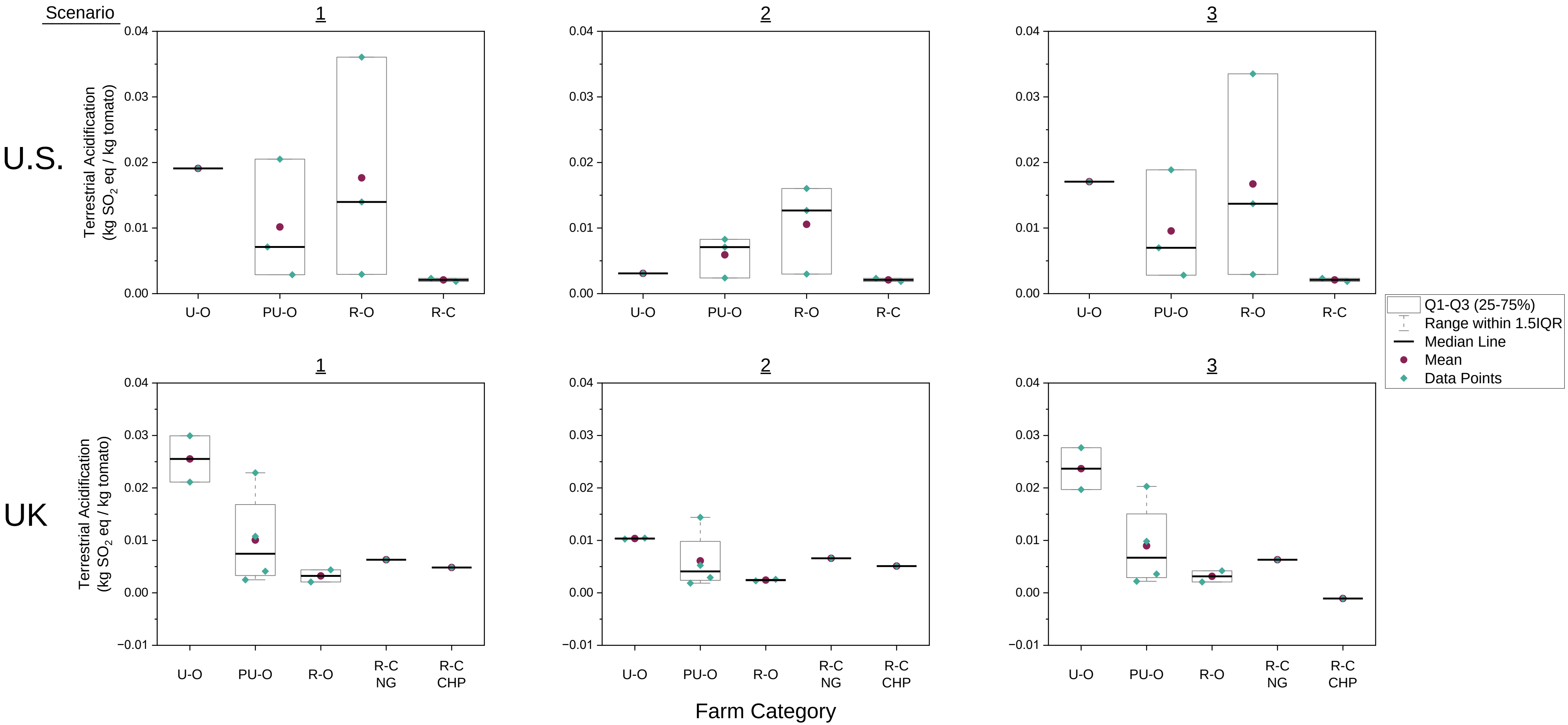

Figure 19 *– Terrestrial acidification per kg tomato transported to final point of sale (kg SO_2 eq / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales differ between U.S. and UK plots.*

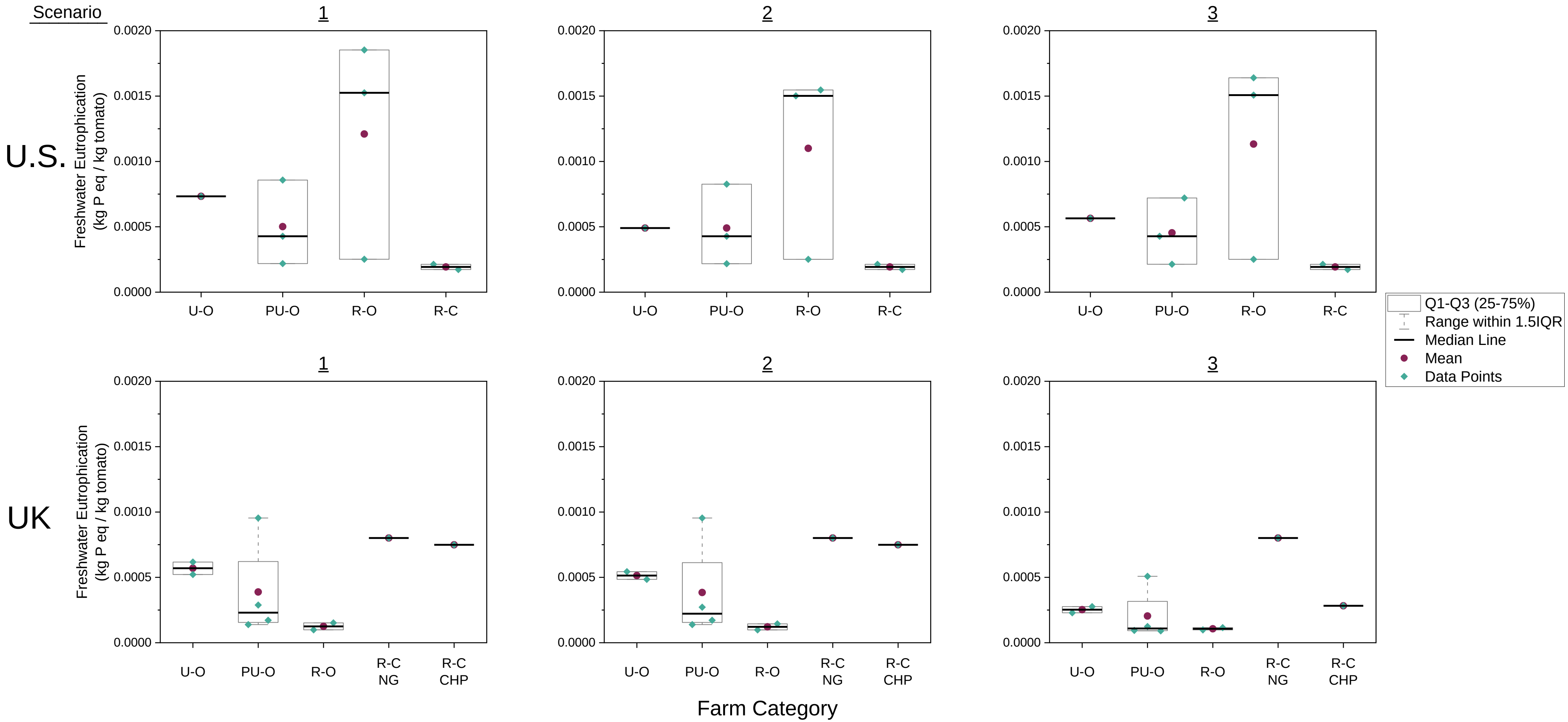

Figure 20 *– Freshwater eutrophication per kg tomato transported to final point of sale (kg P eq / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

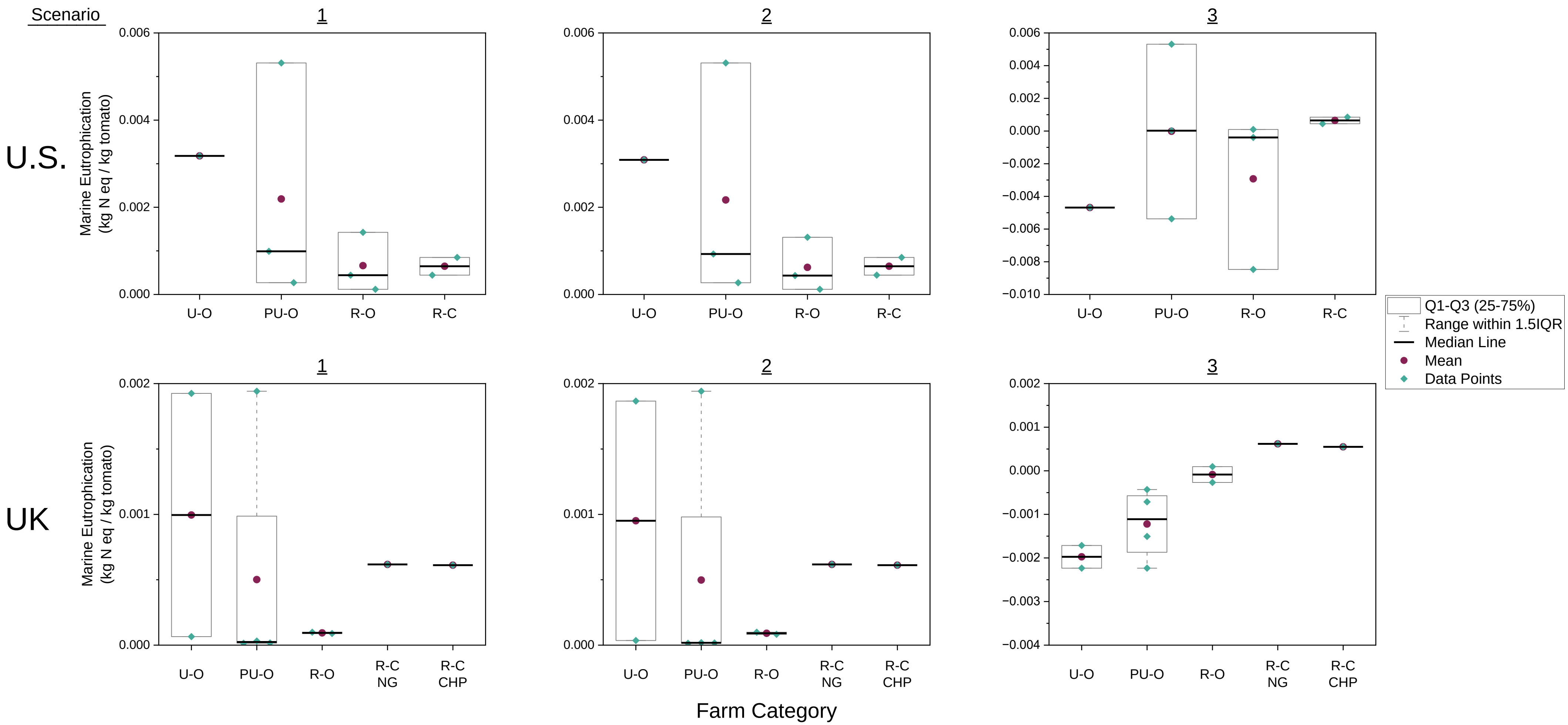

Figure 21 *– Marine eutrophication per kg tomato transported to final point of sale (kg N eq / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Note that axis scales differ between U.S. and UK plots and for Scenario 3 plots due to negative impacts.*

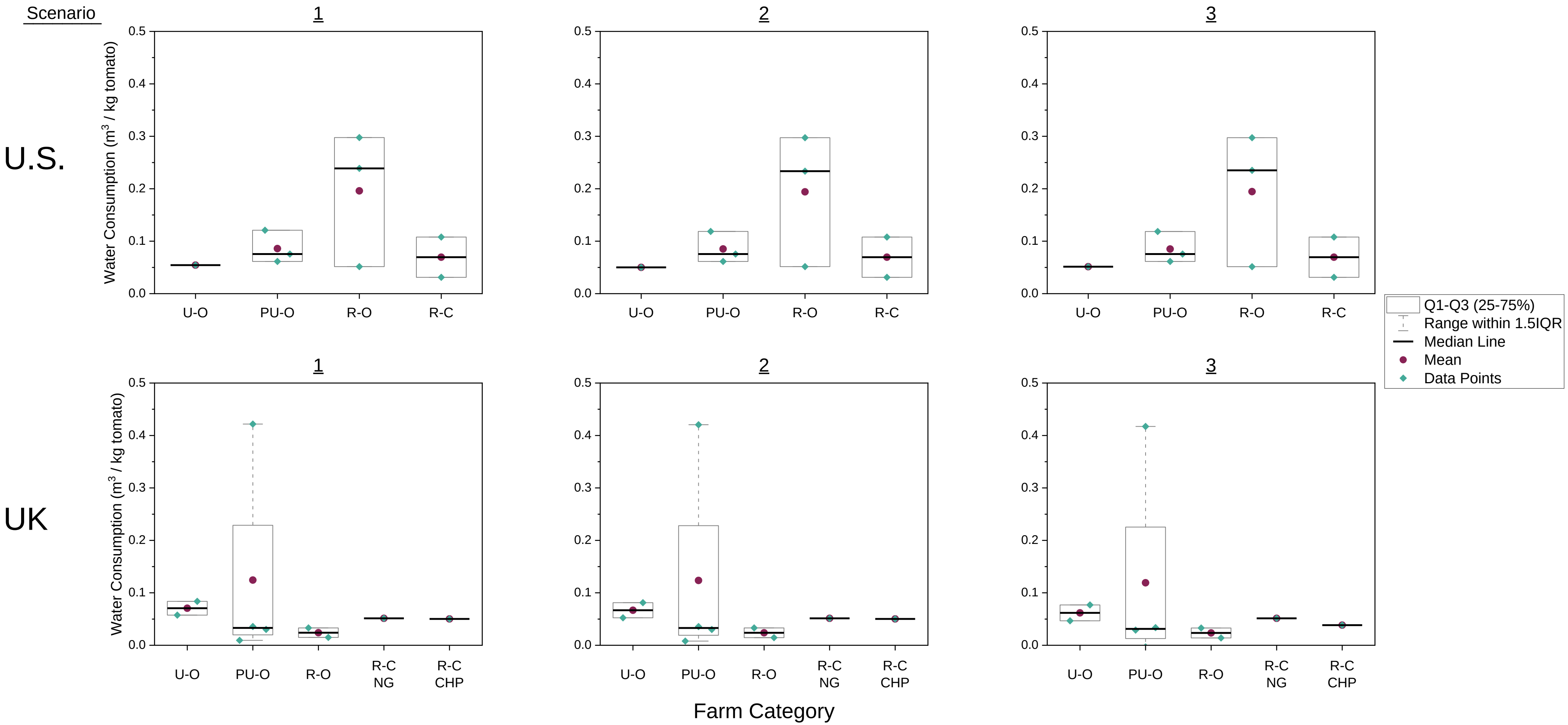

***Figure 22** – Water consumption per kg tomato transported to final point of sale (m^3 / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales are the same for all plots.*

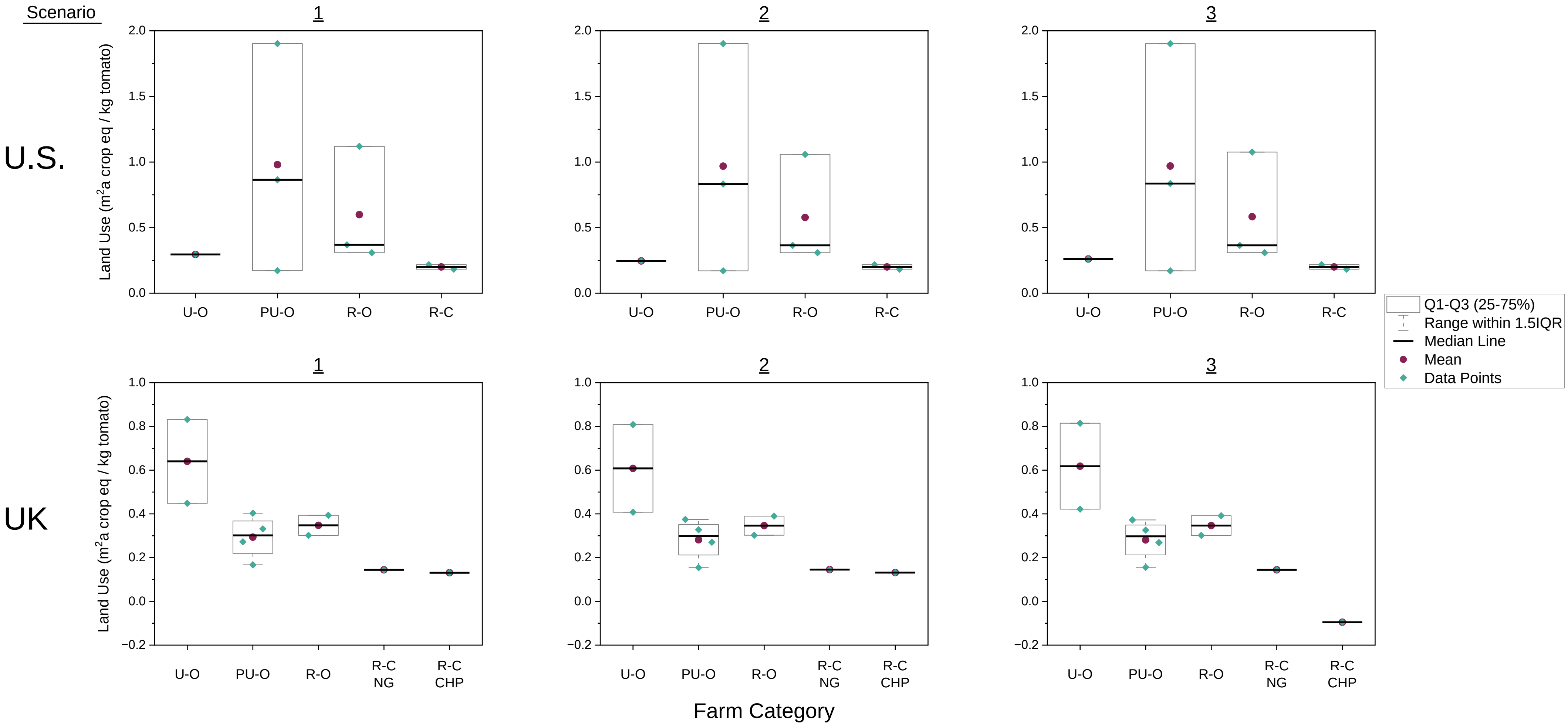

Figure 23 *– Land use per kg tomato transported to final point of sale (m^2a crop eq / kg tomato), plotted per farm category for U.S. (top row) and UK (bottom row) production. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3. Within individual plots, box plots are presented for each farm category, including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C). For UK production only, two conventional production cases are considered based on the energy source used for glasshouse production: either using natural gas and the UK national electricity grid (R-C NG) or combined heat and power (R-C CHP). The box plots include the following data, as also signified in the legend: median (black, bold line), mean (magenta circle), interquartile range (grey box), range within 1.5x the interquartile range (grey whiskers), and individual farm data points (turquoise diamonds). Axis scales differ between U.S. and UK plots.*

3.1.1.2 Scenario comparison

Three different allocation scenarios are assessed in this LCA (see: Section 2.1.5), with results for each scenario included side-by-side in Figure 10-Figure 16 for kale production and Figure 17-Figure 23 for tomato production. Scenario selection does not affect results for every farm or every impact category. Farm results are only affected if they use compost as an input or if they compost crop waste on the farm; additionally, results will also be impacted for farms that utilise CHP (UK conventional tomato production only) within Scenario 3. In both the U.S. and UK, the farms that utilise the most compost are generally urban and peri-urban organic farms. Thus, scenario selection is often seen to impact these farm categories the most. Rural organic farms often also utilise compost, although generally to a lower extent than the more urban farms, so scenario selection usually results in a lower magnitude of change for these farms. Finally, rural conventional farms are not affected by the change in scenarios, except for UK conventional tomato production.

3.1.1.2.1 Scenario 2

The change in results when considering Scenario 2 in comparison to Scenario 1 are discussed first. The impact categories most affected by the re-allocation of composting burdens in Scenario 2 include global warming, fine particulate matter formation, terrestrial acidification, and to a lesser extent, freshwater eutrophication. Changes are also observed for stratospheric ozone depletion, ionizing radiation, and the environmental ecotoxicity categories (Appendix B, Table 75-Table 78), although these are not discussed within this section. Overall, the switch from Scenario 1 to 2 lowers impacts for these categories for the farms that use compost as an input, which are organic farms only, although the magnitude of change depends on the impact category, the amount of compost used on the farm, the type of compost used, and the amount of crop and green waste generated on the farm. On the other hand, the marine eutrophication, water consumption, and land use categories remain mostly unchanged when switching from Scenario 1 to Scenario 2, indicating that the composting process is not a significant driver of impacts for these categories.

The decrease in impacts from Scenario 1 to 2 arises from several factors within the composting process. The amount of compost applied as a fertiliser on farms is generally much higher than the amount that is generated from green waste within the crop lifecycle; thus, the attributed emissions coming from the compost pile itself are reduced when moving from Scenario 1 to Scenario 2. Also, for the industrially-produced composts that are used as fertilisers on some farms, there are additional inputs considered for the composting process. This includes the tractor operations used to turn the compost piles; electricity use; land and water use; infrastructure; and the municipal waste collection for the municipal compost (see Table 41 for the LCI for industrially-produced compost). Thus, for farms that use a high amount of municipal or other industrially-produced composts (e.g., from windrows) as fertilisers, the switch from Scenario 1 to 2 eliminates these additional material and energy inputs from the composting lifecycle, thus resulting in further impact reductions. This is because the composting process for crop green waste that is considered in Scenario 2 happens on the farm, and thus is assumed to include only land use, water use, and the emissions arising from the compost pile, as most small farms do not have significant material or energy inputs for on-farm composting. These are also the only inputs considered for the lifecycle of homemade composts used as fertilisers on farms in Scenario 1 (excluding transport). This difference between compost types is the reason why greater impact changes are observed for farms using industrially-produced composts versus home-produced composts when moving from Scenario 1 to 2.

In particular, for the municipal composting process, the composting emissions themselves only account for 33% of global warming impacts; 12% of fine particulate matter formation impacts; and 53% of terrestrial acidification impacts (per kg compost), whilst for low-input, large-scale windrow composting, these emissions account for 54% of global warming impacts; 63% of fine particulate matter formation impacts; and 91% of terrestrial acidification impacts. For home composting, the composting emissions account for 100% of impacts in these categories, when not considering any transport burdens for off-farm compost feedstocks (e.g., woodchips or manure). Finally, reductions in the freshwater eutrophication category are seen only for farms using industrially-produced composts, as the fugitive emissions from the composting process do not affect this impact category at all; rather, the freshwater eutrophication category is influenced by the electricity, material use, and tractor operations for industrially-produced composts.

When comparing impact categories across Scenario 1 and 2, the largest changes are observed for terrestrial acidification, followed by fine particulate matter formation, global warming potential, and freshwater eutrophication. The variation within and between organic farm categories for these impacts are reduced. Despite these changes, overall trends and outcomes as observed for Scenario 1 mostly remain the same, as all organic farm categories are reduced collectively, although larger reductions are generally seen for the urban and peri-urban farm categories as opposed to the rural organic, due to the higher compost use on more urban farms. Additionally, the rural conventional farm categories are not affected by the switch from Scenario 1 to 2, except for UK conventional tomato production, as this farm composts crop waste on-site. The following paragraphs discuss specific changes in more detail for U.S. and UK kale production first, then U.S. and UK tomato production.

In agreement with the general trend observed across all LCAs, the switch from Scenario 1 to Scenario 2 for U.S. kale production reduces average impact levels for all organic farm categories. The magnitude of these reductions is the least for the rural organic farm category, which sees negligible reductions (<6%) in the category average for global warming, fine particulate matter formation, and freshwater eutrophication, and reductions of approximately 14% for terrestrial acidification. For the peri-urban farm category, the differences are higher, with average impact levels reduced by 7% for freshwater eutrophication, 18% for global warming, 28% for fine particulate matter formation, 53% for terrestrial acidification. The urban farm (n=1), however, shows the greatest changes, with global warming and fine particulate matter formation impacts decreased by approximately 70%, terrestrial acidification impacts by 87%, and freshwater eutrophication by 33% (due to the exclusive use of industrially-produced compost). This indicates that compost use is a main impact driver for this urban farm.

Although in Scenario 1, the urban organic farm has highest impact levels for the previously-mentioned impact categories when compared to other farm category averages, the switch to Scenario 2 decreases the urban farm impact levels to be more in line with other organic category averages. This essentially erases the trend originally seen in Scenario 1, where increasing scale and rurality saw decreased global warming, fine particulate matter formation, and terrestrial acidification impacts, because in Scenario 2 the organic farm category averages are now quite similar. This points to a decrease in variation between all organic farms when moving from Scenario 1 to 2, although one outlier still exists within the peri-urban farm category.

The reductions in variation within farm categories for U.S. kale production is seen most within the peri-urban farm category (n=4), which has the highest variation in Scenario 1. In particular, the switch from Scenario 1 to 2 results in the three peri-urban farms with the lowest impacts displaying much more similar impact levels. The magnitude of the range between these three farms is reduced by about 50% for global warming and fine particulate matter formation and by approximately 80% for terrestrial acidification. However, there is one peri-urban farm that still exists as an outlier, displaying the maximum impact levels for this farm category even though impact levels for this farm were slightly reduced when moving from Scenario 1 to 2. This indicates that there are other factors besides compost driving impacts for this outlier.

For UK kale production, the largest decreases in impact levels are observed for the peri-urban and rural organic farm categories. Unlike for U.S. kale production, the urban farm considered for UK kale production does not use compost as an input and thus is unchanged by the switch from Scenario 1 to 2, as also seen for the UK conventional farm category (n=5). The peri-urban farm category average, however, is reduced by approximately 60% for global warming, 40% for fine particulate matter formation, and 50% for terrestrial acidification. Similar to U.S. kale production, the rural organic category sees less intense changes when moving from Scenario 1 to 2 due to the lower use of compost on these farms. In particular, the global warming impacts are relatively unchanged, whilst differences of approximately 25% are observed for fine particulate matter formation and terrestrial acidification. Freshwater eutrophication is not greatly influenced by the switch from Scenario 1 to 2 for any farm categories, as most farms did not use high amounts of municipal or industrial compost for kale production (as done for tomato production).

As for U.S. kale production, reductions in variation within and between organic farm categories are also observed for UK kale production when moving from Scenario 1 to 2. This is seen especially for the peri-urban farm category, which showed relatively high variation compared to the rural organic (n=2) and conventional farm categories (n=5) in Scenario 1. The peri-urban farms that have the highest impact levels within the category for global warming, fine particulate matter formation, and terrestrial acidification in Scenario 1 see reductions in Scenario 2 that bring impact levels much closer to the farms with the lowest impact levels in the category. These reductions in variation can be quantified through the change in the interquartile range for the peri-urban farm category, which is reduced by approximately 90% for global warming, 60% for fine particulate matter formation, and 95% for terrestrial acidification when moving from Scenario 1 to 2. This therefore indicates that the high impacts in these categories observed for certain peri-urban farms in Scenario 1 are largely driven by compost production.

For U.S. tomato production, the magnitude of changes observed when moving from Scenario 1 to 2 are similar to that observed for kale production. The rural organic (n=3) and peri-urban organic farm category (n=3) averages are decreased by similar proportions, with reductions of approximately 15% for global warming, 21% for fine particulate matter formation, and 40% for terrestrial acidification. For freshwater eutrophication, the peri-urban farm category shows negligible change while the rural organic category average is reduced by 9%. Finally, the urban organic farm (n=1) again shows the greatest magnitude of change, with levels similar to those observed for kale production. This indicates the importance of compost production in driving impacts for both kale and tomato crop lifecycles on this farm.

Again, the switch to Scenario 2 for U.S. tomato production reduces the variation observed among organic farms in Scenario 1. Particularly, the high variation seen in the peri-urban and rural organic farm categories is greatly reduced, and the urban farm impact also decreases to be more similar to other organic farms. These reductions in variations and average impact levels create different trends than those observed for U.S. kale production. Unlike for U.S. kale, where the rural organic farm category often had the lowest average impact levels compared to the other organic farm categories in Scenario 1, for U.S. tomato production the rural organic farm category average was often the highest for many impact categories, with the urban organic farm often similar or close behind. However, in Scenario 2 the urban farm impact levels are decreased to be more in line with or even lower than the peri-urban organic farm category average. Because of this, a clear trend now emerges for U.S. tomato production in Scenario 2, where increasing scale and rurality of organic farms actually results in increasing impact levels for global warming, fine particulate matter formation, terrestrial acidification, and freshwater eutrophication.

For UK tomato production, all farm categories are affected by the switch from Scenario 1 to 2. For the peri-urban (n=4) and rural organic farm categories (n=2), reductions in average impact levels are generally similar to those seen for UK kale production. In particular, changes in the peri-urban farm category average are slightly lower than for kale, with reductions of 54% for global warming, 28% for fine particulate matter formation, and 39% for terrestrial acidification. For the rural organic farm category average, a reduction of 11% is seen for global warming potential, 14% for fine particulate matter formation, and 25% for terrestrial acidification. Changes for freshwater eutrophication are negligible for both peri-urban and rural organic farm category averages, due to the use of primarily home-made composts and manures.

In contrast to all other kale and tomato production cases, for UK tomato production the conventional farm does actually have a slight change in impacts when moving from Scenario 1 to 2, as this farm composts green and crop waste from the tomato plants on site. Thus, since compost is not used as an input on this farm, the inclusion of composting emissions in Scenario 2 for crop waste results in slightly higher impacts when moving from Scenario 1 to 2. However, these increases are essentially negligible (<6%) for global warming, fine particulate matter formation, and terrestrial acidification, and non-existent for freshwater eutrophication, indicating that the composting of crop waste is not a significant contributor to impacts for UK conventional tomato production.

Another difference for UK tomato production in comparison to kale is that the urban organic farm category (n=2) is also impacted when moving from Scenario 1 to 2. This is because both of the urban farms utilised high levels of compost for tomato production, in contrast to the one urban organic farm considered for UK kale production, which did not utilise compost for this crop. The urban farm category average for tomato production is thus reduced by 70% for global warming, 53% for fine particulate matter formation, 60% for terrestrial acidification, and 10% for freshwater eutrophication when moving from Scenario 1 to 2. These reductions bring the urban farm average impact much closer to the other organic farm category averages in Scenario 2, although the urban farm category still has the highest average impacts out of the organic farm categories, as seen in Scenario 1. Thus, the basic outcomes observed in Scenario 1 for these impact categories remain unchanged; decreasing impact with increasing farm scale and rurality is still observed among the organic farm categories. However, as seen with other cases, variation between and within farm categories is generally reduced, seen especially for the urban and peri-urban farm categories.

Collectively, the results for Scenario 2 indicate that the burdens of the composting process for compost inputs are one of the main drivers of impacts on organic farms and of the variation in impact levels seen between organic farms in Scenario 1. This is particularly evident among the urban and peri-urban organic farms, as these tend to use higher amounts of compost as fertiliser in comparison to the more rural organic farms. However, several organic farms still portray high impacts when moving from Scenario 1 to 2, which also suggests that more detail is needed to unpick other impact drivers on organic farms.

3.1.1.2.2 Scenario 3

In Scenario 3, avoided burdens are subtracted from the composting process for compost inputs and from the surplus electricity generated through CHP for UK conventional tomato production. Thus, the switch from Scenario 1 to Scenario 3 always results in reduced impacts for the farms and impact categories affected. As in Scenario 2, the main farms that are affected by the switch to Scenario 3 are the organic farms that use compost as a fertiliser input, although now with the addition of UK conventional tomato production via CHP. A wider range of impact categories are affected in Scenario 3 versus Scenario 2 because of those influenced by the 'avoided burden' processes, namely municipal solid waste (for avoided burdens of compost) and the UK national electricity grid (for avoided burdens of surplus electricity from CHP).

The change in impacts seen for organic farms using compost are discussed first. For this case, ecoinvent v.3.0 municipal waste processes are used to represent avoided burdens from composting. These processes include transport burdens, infrastructure, energy use, and also emissions from the waste treatment process itself, which for the UK is considered as 65% incineration and 35% sanitary landfill and for the U.S. is considered as solely sanitary landfill. The main impact categories affected by the subtraction of avoided burdens from these processes in Scenario 3 include global warming potential (Figure 10 and Figure 17) and marine eutrophication (Figure 14 and Figure 21), out of the impact categories primarily discussed in this results section. Additionally, freshwater ecotoxicity, marine ecotoxicity, human carcinogenic toxicity, and human non-carcinogenic toxicity also show extreme reductions in impacts (over 1000% in some cases), as outlined in Appendix B Table 75 through Table 78, although these impacts are not discussed within this section. All other impact categories are also affected for the farms that have avoided burdens applied, but the differences are less severe and often negligible.

For the impact categories mentioned as being most affected by Scenario 3, the switch from Scenario 1 to 3 will always result in lower impacts than from Scenario 1 to 2. However, for fine particulate matter formation and terrestrial acidification, the switch from Scenario 1 to 2 results in greater impact reductions. For other categories, such as freshwater eutrophication, land use, and water use, changes are often similar and less drastic.

A significant outcome of Scenario 3 is that it actually results in overall negative impacts for certain impact categories. This is seen particularly for both global warming potential and marine eutrophication. In these cases, the switch to Scenario 3 creates an entirely different result outcome from Scenario 1. Generally, the more urban farms that often have higher impacts in Scenario 1 due to high compost use now have the lowest (most negative) emissions in Scenario 3, as they are applied higher benefits in terms of the avoided burdens. Thus, for these impact categories, higher compost use will result in lower impacts, in complete contradiction to Scenario 1. For both global warming and marine eutrophication, the urban farm category has relatively high impacts compared to other farm categories averages in Scenario 1, but in Scenario 3 now has the lowest impacts. This is seen for U.S. and UK tomato production and U.S. kale production, but not for UK kale production, as the urban farm in this case did not use compost as an input.

Similarly, many peri-urban farms also use relatively high amounts of compost and thus show high impact reductions in Scenario 3, often also having overall negative impacts for global warming and marine eutrophication across crop and country lifecycles. Rural organic farm category averages are also reduced, although often to a lesser extent due to the generally lower compost use on these farms, as in Scenario 2. Thus, for both countries and both crops, a basic trend is observed where increasing farm scale and rurality results in increasing global warming and marine eutrophication impacts within Scenario 3. For UK tomato production, this trend is a complete reversal from that observed in Scenario 1. Also, for UK kale production, the trend of increasing impacts with increasing scale is observed but excludes the urban organic farm category, which did not use compost and so is not affected by the scenario change.

The highest magnitude of reductions when moving from Scenario 1 to 3 are generally observed for the urban and peri-urban organic farms in contrast to the rural organic, as often seen in Scenario 2. For U.S. production, the change from Scenario 1 to 3 brings reductions of over 300% for global warming potential and over 200% for marine eutrophication for the urban organic farm, for both tomato and kale production. The U.S. peri-urban organic farm category average for global warming potential is reduced by approximately 120% for both tomatoes and kale; for marine eutrophication, reductions of 100% are seen for tomatoes and 830% for kale. Finally, the U.S. rural organic farm category average is reduced by a much higher proportion for tomato production versus kale, due to the much higher use of compost for tomato production on these farms. In particular, the U.S. rural organic farm average is reduced, for kale and tomatoes respectively, by 31% and 83% for global warming, and by 60% and 541% for marine eutrophication.

For UK kale production, the peri-urban farm category average is reduced by 295% for global warming and 73% for marine eutrophication when moving from Scenario 1 to 3, whilst the rural organic farm category average is reduced by 170% for global warming and 28% for marine eutrophication. The urban farm is unchanged as it does not use compost for kale production. When considering UK tomato production, reductions seen in the urban and peri-urban farm averages are similar, being reduced by approximately 300-350% for global warming and marine eutrophication. Finally, the rural organic farm category average is reduced by 125% for global warming and 190% for marine eutrophication.

The reduction in variation between organic farms is not observed as clearly in Scenario 3 as for Scenario 2, as large variations still exist in Scenario 3. This is because Scenario 2 eliminates or severely reduces the burdens applied to the composting process, while Scenario 3 applies benefits for the composting process (in the form of avoided burdens). Thus, farms that apply relatively high amounts of compost will have relatively low or negative impact levels, in contrast to farms that use lower amounts of compost or no compost at all, thus resulting in high levels of variation between farms.

The use of combined heat and power for energy generation in glasshouse production is also applied avoided burdens in Scenario 3, specifically for the surplus electricity generated. This is relevant only for the case of UK conventional tomato production, and thus the avoided burden is considered as the GB national electricity grid. Using the relevant ecoinvent v.3.0 process, it is seen that the GB electricity grid is dominated by energy production via natural gas (43%), nuclear (22%), and wind (15%), with lower amounts also produced via coal combustion (8%) and CHP using biomass (6%). The impact categories most affected by the avoided burdens of this process, out of those reported in this results section, include fine particulate matter formation (Figure 18), freshwater eutrophication (Figure 20), terrestrial acidification (Figure 19), and land use (Figure 23), with the latter of the two actually portraying negative impacts from the subtraction of avoided burdens. Respectively, reductions of 97%, 62%, 122%, and 173% are seen for these impact categories when moving from Scenario 1 to 3. Reductions in land use impacts are mostly related to the avoided burdens of CHP using biomass (woodchips) within the national electricity grid. Additionally, both ozone formation categories, as well as the ionizing radiation category, also see large reductions from the subtraction of avoided burdens for UK conventional tomato production in Scenario 3, with the latter seeing reductions over 1000% and the emergence of negative impacts (Appendix B, Table 78). On the other hand, global warming potential is less affected by the avoided impacts of surplus electricity, likely due to the relatively high amount of nuclear and renewable energy in the UK national electricity grid.

Although in Scenario 1 conventional tomato production via CHP has some of the highest impacts for fine particulate matter formation, freshwater eutrophication, and terrestrial acidification out of the other UK farm categories, the switch to Scenario 3 results in this category now having some of the lowest impacts. Since these avoided burdens are not applied to the case of conventional production via conventional energy sources, much greater variations are thus observed between the two cases of UK conventional tomato production in Scenario 3. This shows that the replacement of traditional electricity generation with CHP can make more energy-intensive forms of agricultural production, such as hydroponic production in heated glasshouses, a more environmentally attractive option.

Overall, the differences observed in the switch from Scenario 1 to Scenario 3 suggest that results for some impact categories can change dramatically when the benefits of composting and CHP are applied through the subtraction of avoided burdens. For organic farms, the results are completely reversed from Scenario 1 to 3 in many cases, as farms using higher amounts of compost will now have some of the lowest impacts, sometimes even negative. Negative impacts are also seen in some categories for UK hydroponic tomato production using CHP, suggesting that this energy-intensive production method can become a more sustainable option if different energy sources are considered. The sensitivity of LCIA results to scenario selection, particularly in regard to compost and CHP allocation, implies that the allocation method can largely shape overall conclusions achieved within LCAs and that the exploration of this is critical. Additionally, the fact that high variation is observed among and within farm categories across scenarios for certain impacts, and that outliers still exist across scenarios, points to the fact that overall conclusions cannot be boiled down to a singular farm category and that specific impact drivers need to be identified on an individual farm level. This information is thus presented in the following section.

3.1.2 Results for individual farms and contributing processes

In order to identify the specific processes contributing to differences in impacts between farms, individual farm impacts per crop type and country have been presented in Figure 24 through Figure 51. Results have been presented as column charts with both actual impact values and with contributing processes displayed as a percentage out of 100, in order to easily identify the magnitude and contribution of impacts. Contributing processes are represented by different colours within the column, with explanations of each process category provided in Table 63. Finally, results are also displayed for each allocation scenario side-by-side.

***Table 63** – Process contribution categories for LCIA*

Contributing process	Description
Seedlings	Burdens of producing seedlings, either on the farm or at a nursery. This includes all material and resource flows (e.g., energy, fertilisers, growing media, germination trays, etc.).
Direct land use, cultivation	Land used for cultivation of the crop on the farm; this is only relevant for the 'land use' impact category
Direct water use, cultivation	Water used for cultivation of the crop on the farm, including either tap water or water from ponds and boreholes; this is only relevant for the 'water consumption' impact category.
Compost production & transport	Burdens of producing compost, for compost inputs, including composting emissions (emissions from decomposition of organic waste in the compost pile) and any energy, material, and infrastructure inputs associated with compost production (see: Section 2.1.7.6.3), as well as packaging and transport of the compost to the farm. For Scenario 2, this will include only burdens from compost transport and any burdens to produce materials purchased for composting (e.g., hay, if purchased). For Scenario 3, this also includes avoided burdens from the subtracted municipal solid waste process.
Composting crop waste	Burdens from the composting process for crop and green waste that is generated from the crop lifecycle of interest on farms. Includes emissions from the composting process and water use. This is only relevant to Scenario 2.
Other fertiliser production	Burdens from the production of fertilisers other than compost, which are used for cultivation, including NPK fertilisers for conventional farms and other organic fertilisers used on organic farms. Includes packaging, packaging waste, and transport.
Agricultural emissions	Direct agricultural emissions as discussed in Section 2.1.9, which come from fertiliser application and incorporation of crop residues into the soil. This includes nitrate leaching emissions calculated based on total N flux (all N inputs and outputs); uptake of CO_2 into the biomass of the harvested crop; and all direct emissions to air from fertiliser application and incorporation of main crop residues into the soil, but not those from cover crops, which are included separately.
Pesticide production & emissions	Burdens associated with pesticide production, packaging, and transport, and the emissions from application as discussed in Section 2.1.9.11. This includes organic and biological pesticides.
Cover crops	Burdens from growing cover crops, including seed inputs and land use. This also includes direct N_2O emissions to air associated with cover crop incorporation, but not N_2O and nitrate emissions associated with leaching, which are accounted for within the agricultural emissions category (due to the need to calculate based on total N flux).
Cultivation material use	Materials used for cultivation, including: irrigation materials (e.g., drip tape), plastic mulches and fleece, hoops for low tunnels, plastic crates for harvesting, trellising materials (for tomatoes only), and growing media (for hydroponic farms only).
Cultivation machinery & operations	Burdens of producing the machinery used farms, such as tractors and implements, as well as the fuel use and emissions associated with operating this machinery. Also includes burdens for vehicles operated on the farm (e.g., for moving in between fields).
Polytunnels & glasshouses	Material burdens of polytunnel / glasshouse infrastructure. Does not include land use, which is included in the 'direct land use for cultivation' category.
Other farm infrastructure	Burdens from other infrastructure used for the crop lifecycle, such as sheds and packhouses.
Cultivation energy use	Burdens from energy use for cultivation, including electricity and heat, such as that used in polytunnels or for irrigation. Also includes energy and infrastructural burdens of using tap water, but not the water flow. For Scenario 3, this process will also include avoided burdens subtracted from the surplus electricity produced by CHP (for UK conventional tomato production only).
Direct water use, processing	Water used for the processing / packaging stage, e.g., for washing crops or misting in cold stores. This includes either tap water or water from ponds and boreholes and is only relevant for the 'water consumption' impact category.
Packhouse operations	Includes burdens associated with processing, packaging, and storage of crops, such as energy use, fuel use (e.g., for forklifts), and any waste burdens associated with the crop. Also includes energy and infrastructural burdens of using tap water use, but not the water flow. This does not include packhouse infrastructure, which is included within 'other farm infrastructure.'
Packaging & transport materials	Material inputs used for packaging and transporting the crop, such as cardboard boxes and plastic bags and films, as well as crates and pallets used for transport.
Final transport & distribution centres	Burdens of transporting the crop to the final point of sale, including vehicles and fuel use. This also includes burdens associated with storage in distribution centres (for conventional farms only).

Results are presented in the following sections per country and crop type, then discussed per impact category. The global warming potential, fine particulate matter formation, and terrestrial acidification impact categories are discussed collectively because they are largely driven by similar processes, and thus, similar trends between farms are observed. The main contributing processes for these impact categories include: the production of compost and other fertilisers; direct agricultural emissions from the application of fertilisers and incorporation of cover crops and crop residues; cultivation machinery and operations, mainly from fuel combustion; energy uses during cultivation and packhouse operations, especially for cold storage of crops; the use of polytunnels and other farm infrastructure; and the transport of the crop to the final point of sale. The main difference is that the contributions from agricultural emissions and cultivation machinery and operations are higher for fine particulate matter formation and for terrestrial acidification than for global warming potential. For fine particulate matter formation, this is because a wider array of N-based agricultural emissions contribute to this impact category (e.g., NH_3, NO_x, and NO_3^-) in contrast to global warming (just N_2O), as seen in Table 62. For terrestrial acidification, the NH_3 emissions associated with compost production and fertiliser application (agricultural emissions) and the SO_2 emissions associated with fuel combustion from cultivation operations cause these processes to all play larger roles in impact contribution. On the other hand, agricultural emissions are not a major contributor of global warming impacts for most organic farms, as emissions from fertilisers and crop residues are normally offset by the biogenic CO_2 accounted for in crop biomass exported off the farm, in some cases causing slightly negative contributions from this process.

For other impact categories, results are discussed separately as the trends observed between farms for global warming potential, fine particulate matter formation, and terrestrial acidification tend to change. This is because the main contributing processes differ for these categories, and compost production in particular is no longer a major contributor. For freshwater eutrophication, the contributing processes vary widely between crops and countries, with impacts generally driven to a higher extent by agricultural emissions, material use (in both cultivation and packaging), infrastructure, and energy use (for U.S. farms) than for the previously discussed impact categories. Marine eutrophication is driven almost exclusively by agricultural emissions, mainly from nitrate leaching impacts. This is seen for all farms except those that grow the crop exclusively indoors using drip irrigation; in this case, nitrate leaching is assumed to be null, following IPCC (2019) guidelines (see: Section 2.1.9.5 for more detail). Water consumption is driven mainly by direct uses for cultivation (e.g., irrigation and spraying pesticides) and processing, and, to a smaller extent, by fertiliser production. Finally, land use is driven by mainly by direct uses in cultivation and for cover crops.

In the following sections, individual contributions will be discussed in more detail for each country and crop lifecycle, also considering differences between allocation scenarios.

3.1.2.1 U.S. Kale

Figure 24 through Figure 30 display global warming, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, water consumption, and land use impacts, respectively, across individual farms for U.S. kale production. These figures also depict the individual processes contributing to impacts on each farm. The trends observed between farms, and the main processes contributing to these trends, are discussed in the following sections.

3.1.2.1.1 Global warming, fine particulate matter formation, & terrestrial acidification

The major contributing processes to global warming potential (Figure 24), fine particulate matter formation (Figure 25), and terrestrial acidification (Figure 26) for U.S. kale production primarily include: compost and other fertiliser production, agricultural emissions, polytunnels and other infrastructure, cultivation energy use, packhouse operations, and final transport. Trends seen for global warming potential, fine particulate matter formation, and terrestrial acidification are similar between impacts categories and thus will be collectively discussed first.

For Scenario 1, the highest impact levels for these categories are generally observed on organic farms that use compost as the main fertility input for kale production, including U-O-1, PU-O-1, PU-O-2, and R-O-1. U-O-1 and PU-O-1 apply the highest amounts of compost (at least 2-3x the amounts applied on other farms), and for these farms, compost production is the single highest impact contributor, accounting for at least 60% of global warming and fine particulate matter formation impacts and over 80% of terrestrial acidification impacts. Large variation in impact levels is observed between organic farms using compost as an input and those that use minimal or no compost, suggesting that compost use (and associated production impacts) is one of the main drivers of variation observed between organic farms, as previously discussed when comparing farm category averages across allocation scenarios (see: Section 3.1.1.2).

Indeed, the lowest impacts among organic farms are observed on the peri-urban farms that use very low amounts of compost as a fertiliser input (PU-O-3) or that use no compost at all (PU-O-4). These farms display impact levels similar to or even lower than the conventional farm impact level. Thus, although the conventional farm tended to have the lowest impacts across most categories in comparison to other organic farm category averages (see: Section 3.1.1.1.1), when comparing individual farms it can be seen that PU-O-3 actually has the lowest overall impact levels. This farm displays impact levels at least half the value of the conventional farm for global warming, fine particulate matter formation, and terrestrial acidification.

The reasons for the relatively low impacts seen for PU-O-3 and PU-O-4 are driven by different reasons. PU-O-3 utilises low-input production practices, growing outside using very low compost inputs (comprised mainly of horse manure), minimal material use, and no irrigation. In addition, all produce is sold on-site through a farm shop and restaurant, and thus this farm has no contributions to impact from final transport. However, the trade-off of these low-input growing practices is a sacrifice in crop output, seen as this farm achieved a yield of only 0.75 kg m^{-2} in comparison to the organic average yield of 1.1 kg m^{-2} and the conventional yield of 2.1 kg m^{-2}. This low yield was largely a result of issues with urban soil conditions (compaction), low soil fertility, and low demand from customers for this crop, resulting in this farm having the highest total crop waste seen across U.S. kale farms (75%).

On the other hand, PU-O-4 also has relatively low impact levels, similar to the conventional farm for global warming potential and fine particulate matter formation and actually lower for terrestrial acidification, but also had one of the highest yields out of all organic farms (1.6 kg m^{-2}). This farm grows crops across neighbours' yards in a suburban area. They applied several different organic fertilisers (e.g., seaweed-based fertilisers, but no compost), utilised irrigation for crop production, and employed several crop protection strategies during growing, such as using fleeces, low tunnels, leaf mulch, and flame weeding. This suggests that the efficient use of fertilisers other than compost and the employment of crop protection strategies can be important for reducing waste, increasing yield, and achieving relatively low global warming, fine particulate matter formation, and terrestrial acidification impacts.

Compost and fertiliser production are not major impact contributors for PU-O-3 and PU-O-4, as they are for other organic farms, and thus the main factors driving impacts on these farms include seedling production, energy use, and infrastructure. The contributions of these factors in actual terms are similar to that observed on other organic farms, but in percent value contributions are higher due to the relatively low overall impact levels on these farms and the lack of other contributing fertiliser-related processes. Energy use in particular is the highest contributor for fine particulate matter formation for PU-O-4, mainly from the energy used for cold storage of kale (counted for within the packhouse operations category).

Although the peri-urban farm category includes farms with some of the lowest impacts overall, it also has farms with some of the highest impacts. This is seen for PU-O-2, the farm that is seen to be main outlier across these impact categories. This farm has a global warming potential of nearly 12 kg CO_2 eq (kg kale)$^{-1}$ in Scenario 1, while all other farms have global warming potentials <6 kg CO_2 eq (kg kale)$^{-1}$. It becomes clear that this farm skewed the average impact level for the peri-urban farm category within the averaged results (see: Section 3.1.1.1.1), which is why the rural organic farm category average was often lower than the peri-urban average (e.g., Figure 10-Figure 12), despite the peri-urban farm category having some of the farms with the lowest individual impacts. This points to the high variation observed between farms in the peri-urban category, and organic farms in general, and thus the importance of uncovering impacts on an individual farm scale.

There are three main sources of the high impacts observed for PU-O-2. Although this farm did have a relatively low yield (0.72 kg m^{-2}), this yield is not dissimilar to other organic farms that were able to achieve lower impacts (e.g., PU-O-3 and R-O-2), and thus this cannot fully explain the high impacts observed. In addition, although this farm sees a high contribution from compost production (approximately 20-50% across impact categories), PU-O-2 still emerges as an outlier in Scenario 2, where compost production burdens are removed for compost inputs. This signals that other factors are driving its emergence as an outlier, which are mainly related to instances of inefficient transport operated by the farm. Like PU-O-4, PU-O-2 operates a model using portions of land across several yards within one neighbourhood. However, they have one larger site located relatively far away from the others (53 km one way). Although this may seem to be an important site for the farm, from an environmental impact standpoint operating this site is largely inefficient, as a relatively small amount of produce is grown there whilst requiring a high transport burden. This burden of transport between farm sites is captured within the cultivation machinery and operations category, which is seen to account for approximately 15-25% of impacts across categories for PU-O-2. The final transport of kale accounts for the last major contribution to total impact, constituting 20-40% across impact categories. Although PU-O-2 distributes mainly to farmers' markets located no more than 15 km away, they also sell a small portion of produce to another farm's CSA, located approximately 58 km away. The burden of transporting a relatively small amount of produce to this farm using a passenger car contributes to high transport impacts.

Significant contributions from the final transport of crops are also observed for the rural organic farms. Final transport accounts for 10-30% of impact across these farms for global warming potential and 5-20% for fine particulate matter formation and terrestrial acidification, compared to <5% across these impact categories for other organic farms (excluding PU-O-2). The rural organic farms also display much higher contributions from final transport in real terms (and similar in percent terms) than the rural conventional farm, despite the fact that kale travels much longer distances to the final point of sale from the conventional farm (1,280 km on average) in comparison to the organic ones (<200 km). The reason for these high contributions from final transport for the rural organic farms is again because of the relatively inefficient modes of transport used, especially in comparison to conventional production. The conventional farm transports the maximum amount of crop each time using fuel-efficient lorries; in contrast, the organic farms transport only the amount of produce that will be sold at farmers' markets or to restaurants in the city using mainly diesel or petrol vans. This points to the importance of final transport impacts for smaller-scale agricultural production in comparison to larger-scale production, which are embedded in a more logistically-efficient food supply chain.

Other impact drivers for rural organic farms are more variable. As these farms did not use high amounts of compost as fertiliser inputs, impacts are driven more by the production of other types of fertilisers and by agricultural emissions. For farms that grew a significant portion of kale inside (>50%), the use of polytunnels is also a major contributor of impacts, as seen for R-O-2 and R-O-3, where this contributes to approximately 20% of impacts across categories. Energy use during cultivation (mainly from irrigation) and during packhouse operations (mainly from cold storage) are also major contributors, especially for fine particulate matter formation, where these collectively comprise 30-50% of total impacts. This contribution is mainly driven by the use of coal power within the southeast U.S. electricity profile.

Agricultural emissions are a more significant driver for terrestrial acidification impacts, constituting approximately 10-25% of impacts on rural organic farms. It can be seen that the contributions from fertiliser production and agricultural emissions contribute much less (in real terms) to terrestrial acidification impacts than does the process of compost production on other organic farms (e.g., U-O-1, PU-O-1, and PU-O-2), making the use of alternative organic fertilisers more attractive for these impact categories in Scenario 1. Finally, for terrestrial acidification R-O-2 emerges as another farm with lower impacts than the conventional farm, not seen for the global warming or fine particulate matter formation impact categories. This is mainly due to the lower contribution from infrastructure and energy use within packhouse operations to terrestrial acidification, which drove higher impacts for this farm in the other categories.

The main impact drivers for the conventional farm include the production of fertilisers and associated agricultural emissions, which collectively account for approximately 35% of total global warming impact, 60% of fine particulate matter formation impact, and 80% of terrestrial acidification impact. In particular, terrestrial acidification impacts for the conventional farm are relatively high in comparison to levels seen for the other impact categories, with impact levels more similar to the other rural organic farms. This is because of the much larger contribution (in both actual terms and percent) from agricultural emissions to terrestrial acidification, which alone accounts for nearly 70% of total impact. The production of pesticides and packaging materials also contribute more to impact for the conventional farm than seen for the organic farms, albeit not as significant an amount as fertiliser production and agricultural emissions. Final distribution accounts for approximately 5-15% across impact categories, although in real terms the impact contribution is much less than seen for the rural organic farms.

When switching to other scenarios, impacts are reduced considerably for some of the organic farms with the highest impacts. In Scenario 2, where compost production burdens for compost inputs are excluded, the relatively high impacts seen for U-O-1, PU-O-1, and R-O-1 due to compost use are largely diminished. Although burdens from composting crop waste are counted in Scenario 2, this always contributes less than the burdens for compost inputs for these farms. In particular, switching to Scenario 2 results in PU-O-1 achieving impact levels lower than the conventional farm for all three impact categories; in this case, three out of the four peri-urban farms have the lowest overall impact levels. PU-O-2 is still an outlier in the data, with much higher impacts than any other farm (at least double the next highest farm), albeit lower than in Scenario 1. U-O-1 also sees a significant reduction in impacts, with impact levels more similar to the rural organic farms and also lower than the rural conventional farm for terrestrial acidification. Indeed, in Scenario 2, all but two organic farms have lower terrestrial acidification impact levels than the conventional farm, showing the importance of compost production in driving impacts for this category.

In Scenario 3, the farms with the highest levels of compost inputs and thus some of the highest impacts now have some of the lowest global warming potentials, due to the avoided burdens attributed to compost inputs. In particular, this results in U-O-1, PU-O-1, and even PU-O-2 all having overall negative global warming potentials, thus becoming the lowest impacting farms for global warming. However, the avoided burdens applied in Scenario 3 are less impactful for the fine particulate matter formation and terrestrial acidification categories. For these cases, impacts for farms using compost are slightly lowered in comparison to Scenario 1, but overall trends are unchanged.

3.1.2.1.2 Freshwater eutrophication

For freshwater eutrophication, trends observed in the previous three impact categories change slightly due to the higher contributions from different processes (Figure 27). In particular, freshwater eutrophication for U.S. kale production is driven mainly by energy uses because of coal power in the southeast U.S. electricity grid (approximately 30%) and the phosphorus and phosphate emissions associated with this. Compost production and other fertiliser production are also seen to be major contributors for certain farms, but this is largely due to the energy uses associated with these processes. Finally, agricultural emissions are also a major driver due to the P leaching emissions associated with compost and fertiliser application.

For freshwater eutrophication, all organic farms have relatively high impact levels in comparison to the conventional farm. The conventional farm is able to achieve relatively low freshwater eutrophication impacts because of the generally low energy use per kg kale, achieved mainly due to the more efficient economies of scale and short storage time of the crop before shipment (thus requiring minimal cold storage). Only PU-O-3 is seen to have lower impact levels than the conventional farm, due largely to the minimal fertiliser use on this farm, and thus the minimal contribution from fertiliser production and agricultural emissions in comparison to other farms. In fact, PU-O-3 has no contributions from agricultural emissions as this farm applies no P fertilisers. Instead, impacts for PU-O-3 are driven almost entirely by seedling production, infrastructure, and energy use for crop storage. PU-O-4 also has low fertiliser use and no compost use, which is why this farm also had relatively low impacts for the previous three impact categories, but for freshwater eutrophication, the high contribution from energy use during seedling production (for heating), cultivation, and cold storage drives impacts higher than the conventional farm.

For the farms using the highest levels of compost (U-O-1 and PU-O-1), compost production and agricultural emissions (from estimated P leaching) are the main drivers of freshwater eutrophication impact. These collectively comprise approximately 70% of total impacts, with the rest driven mainly by cultivation energy use. The particularly high compost application for the urban farm leads to this farm having one of the highest impact levels across this impact category. PU-O-2, again the farm with the high impact level (as seen for the previous three categories), also has a significant contribution from compost production and agricultural emissions (25%), but also sees additional contributions from cultivation operations and final transport. This is due to the emissions associated with fuel combustion in vehicles during transport between farm sites and to final points of sale. However, unlike for the previous three categories, PU-O-2 is no longer an outlier for these results, as similarly high impacts are observed for U-O-1 and R-O-3.

Energy use from cultivation and within packhouse operations dominates freshwater eutrophication impacts for the rural organic farms, constituting 40-50% of total impacts, with the rest driven mainly by fertiliser production, agricultural emissions, and infrastructure. Interestingly, R-O-3 also sees a significant contribution from pesticide production and emissions (30%), due to the emissions associated with the production of insecticidal soap and wastewater generated during this process. However, this pesticide process presents a degree of uncertainty, as this has been modelled using soap production as an approximate. It can be seen that this pesticide production, along with cultivation energy use, drives the relatively high freshwater eutrophication impact for R-O-3, similar to levels seen on U-O-1.

When comparing across scenarios, no major differences in trends are observed because compost production is not a major driver of freshwater eutrophication impacts. Impacts are slightly lowered for the farms using compost in both Scenario 2 and 3, but this does not change overall results.

3.1.2.1.3 Marine eutrophication

For marine eutrophication, impacts are driven almost exclusively by agricultural emissions, in particular from nitrate leaching (Figure 28). Because of the relatively high N input from synthetic fertilisers applied on the conventional farm, almost all organic farms are seen to have lower marine eutrophication impacts. The exception is U-O-1, which actually applied higher amounts of total N (mainly from compost) than the conventional farm, and also R-O-1, which saw additional contributions from fertiliser production and seed production for cover crops. The lowest impacts are again observed for PU-O-3 and PU-O-4, as for most other previously discussed impact categories. PU-O-3 has a relatively low contribution from agricultural emissions due to the minimal N inputs applied (from a very low compost application). PU-O-4, on the other hand, sees no contribution from agricultural emissions because of a very low N input per kg kale produced, which resulted in a negative N flux within nitrate leaching calculations (see: Equation 11 for calculation information).

Compost production plays a negligible role in impacts in comparison to agricultural emissions, as none of the fugitive emissions generated from the compost pile actually contribute to marine eutrophication. Thus, no major differences in impacts are observed when comparing Scenarios 1 and 2. However, the avoided burdens from the municipal solid waste stream applied to compost inputs results in relatively high negative contributions from the composting process in Scenario 3. This results in the farms using the most compost (U-O-1, PU-O-1, and PU-O-2) all having overall negative marine eutrophication impacts. It should be noted that avoided burdens are not applied to PU-O-3, because this farm's compost components would not have alternatively gone to the municipal waste stream (hay and manure).

3.1.2.1.4 Water consumption

Water consumption is driven primarily by water use during cultivation (mainly for irrigation), constituting >60% of impacts for nearly all farms (Figure 29). The exceptions are PU-O-2, which used almost as much water for washing kale as for irrigating, and PU-O-3, which did not irrigate kale at all, thus having almost all water consumption coming from washing kale. Water consumption impact levels are generally similar among farms, in the range of 0.08-0.22 m^3 (kg kale)$^{-1}$, and variation does not necessarily seem correlated with farm category. The exception is R-O-3, which emerges as an outlier with a water consumption of 0.54 m^3 (kg kale)$^{-1}$. This is due to the fairly high irrigation levels employed by this farm compared to others, with an additional contribution also coming from water use in pesticide production (9%). As compost production is not a major contributor to this impact category, results are largely unchanged between scenarios.

3.1.2.1.5 Land use

Land use for U.S. kale production is driven primarily by direct land use for cultivation, which is correlated to yield (Figure 30). Additional contributions come from the land used when growing cover crops prior to kale production, as seen for U-O-1, R-O-1, and R-C-3. Higher land uses from cover crops are attributed to those that are grown for longer periods of time before kale production.

As expected, the farms with the highest yields have the lowest land uses. In particular, PU-O-1 has the lowest overall land use and the highest yield out of all organic farms (2.0 kg m^{-2}). This farm used a relatively intensive production strategy, growing kale on a small-scale with moderate inputs of compost and manure. The conventional farm (R-C-3) has the second lowest land use, although the highest yield (2.1 kg m^{-2}), with additional contributions seen mainly from the use of cover crops. Relatively low land uses are also seen for most other urban and peri-urban farms, although none lower than the conventional farm. The exception is PU-O-2, which has relatively high land use impact compared to other urban and peri-urban farms, due in part to additional contributions from infrastructure (36%). This is mostly attributed to the wood used in the household garages where this farm stores its crops, which generate higher land use impacts in comparison to larger shed and barn structures seen on other farms that have lower material use per area. In addition, although the urban farm had a similar amount of direct land use for cultivation in comparison to the conventional farm (in actual terms), additional contributions from compost production and cover crops drove the impacts for this farm higher.

In contrast, the rural organic farms generally exhibit the highest land uses. This is driven in part by their relatively low yields in comparison to some of the smaller-scale, more intensive urban and peri-urban farms, but also by other land use contributions. The use of polytunnels contributes partly to these impacts, mainly from the land use associated with the wood used in the tunnels. Fertiliser production also contributes to higher impacts for R-O-1 and R-O-2, specifically due to use of peat moss and okraseed meal for the prior and alfalfa meal for the latter, all which have relatively high land uses for production.

Comparing across scenarios, trends do not change because compost production is not a significant contributor to land use in comparison to other direct land uses for cultivation or cover crops. Land use is only slightly reduced for farms using compost in Scenario 2, and even less so in Scenario 3.

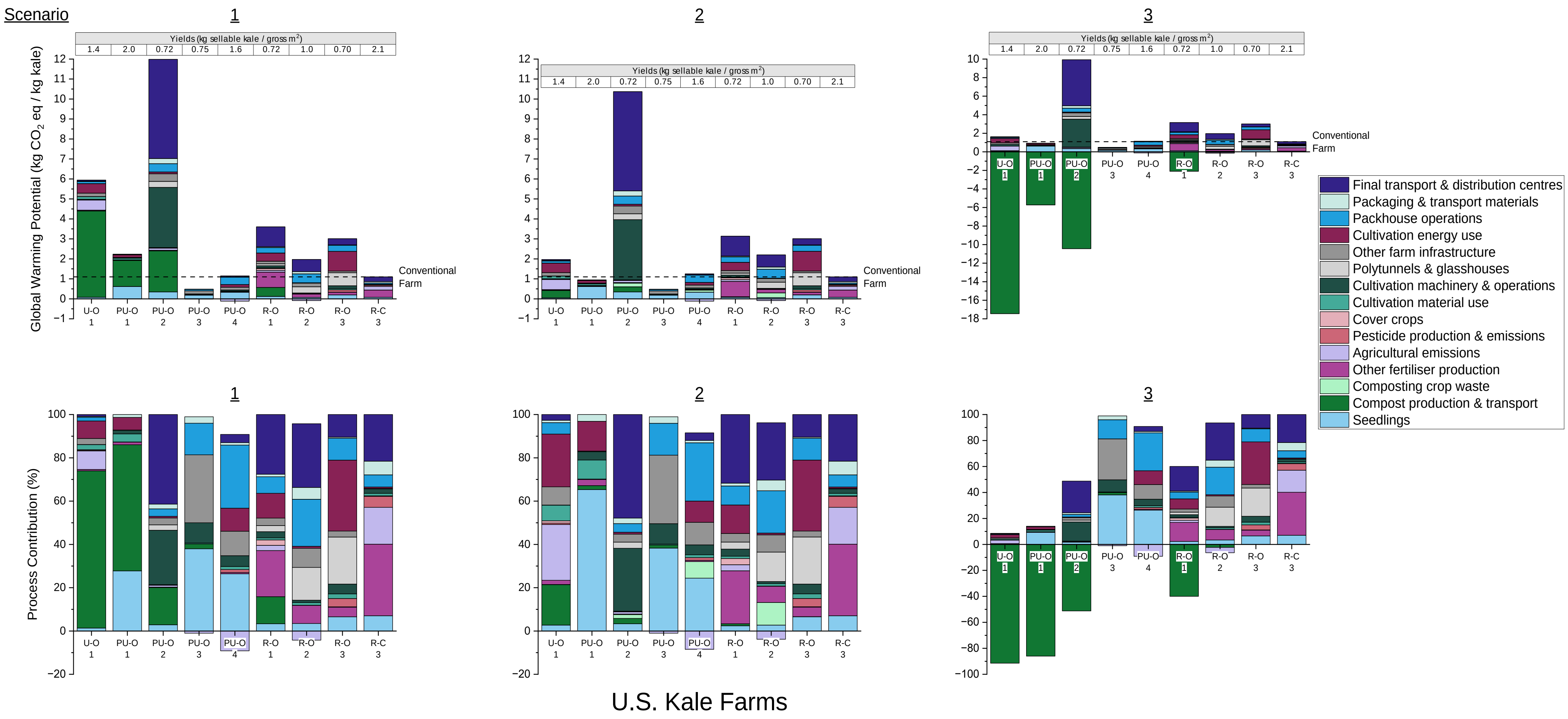

Figure 24 *– Global warming potential and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg CO_2 eq per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

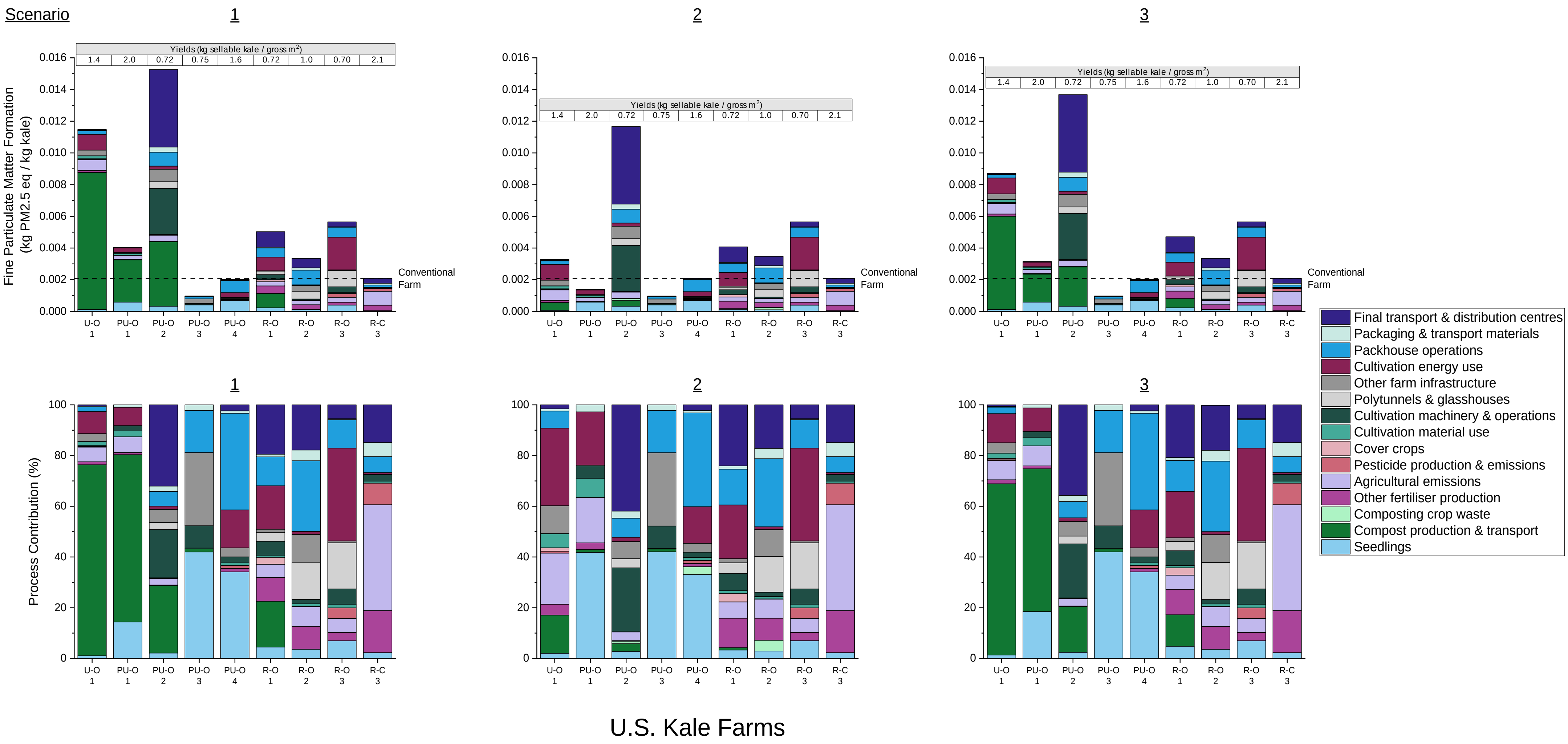

Figure 25 *- Fine particulate matter formation impacts and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg PM2.5 eq per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

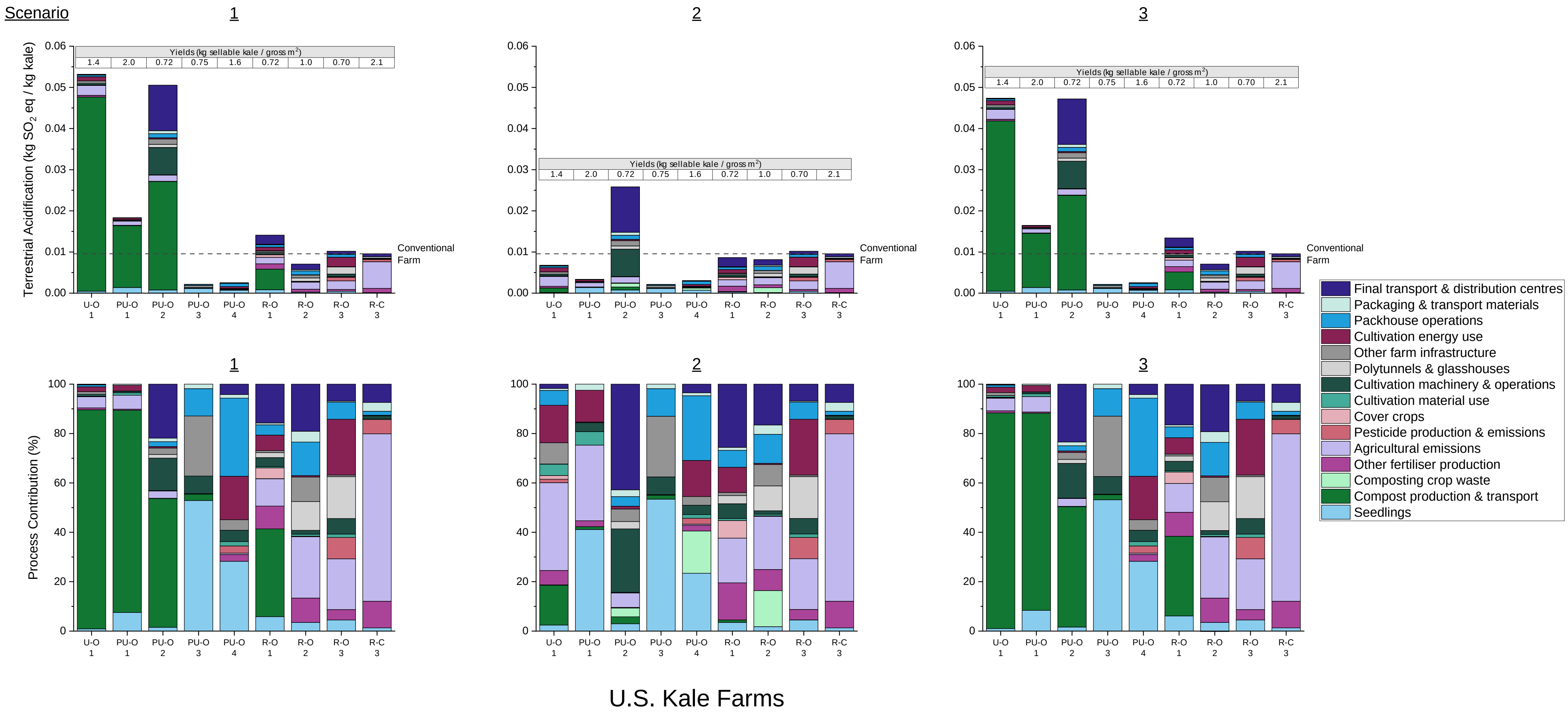

Figure 26 *– Terrestrial acidification impacts and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg SO_2 eq per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

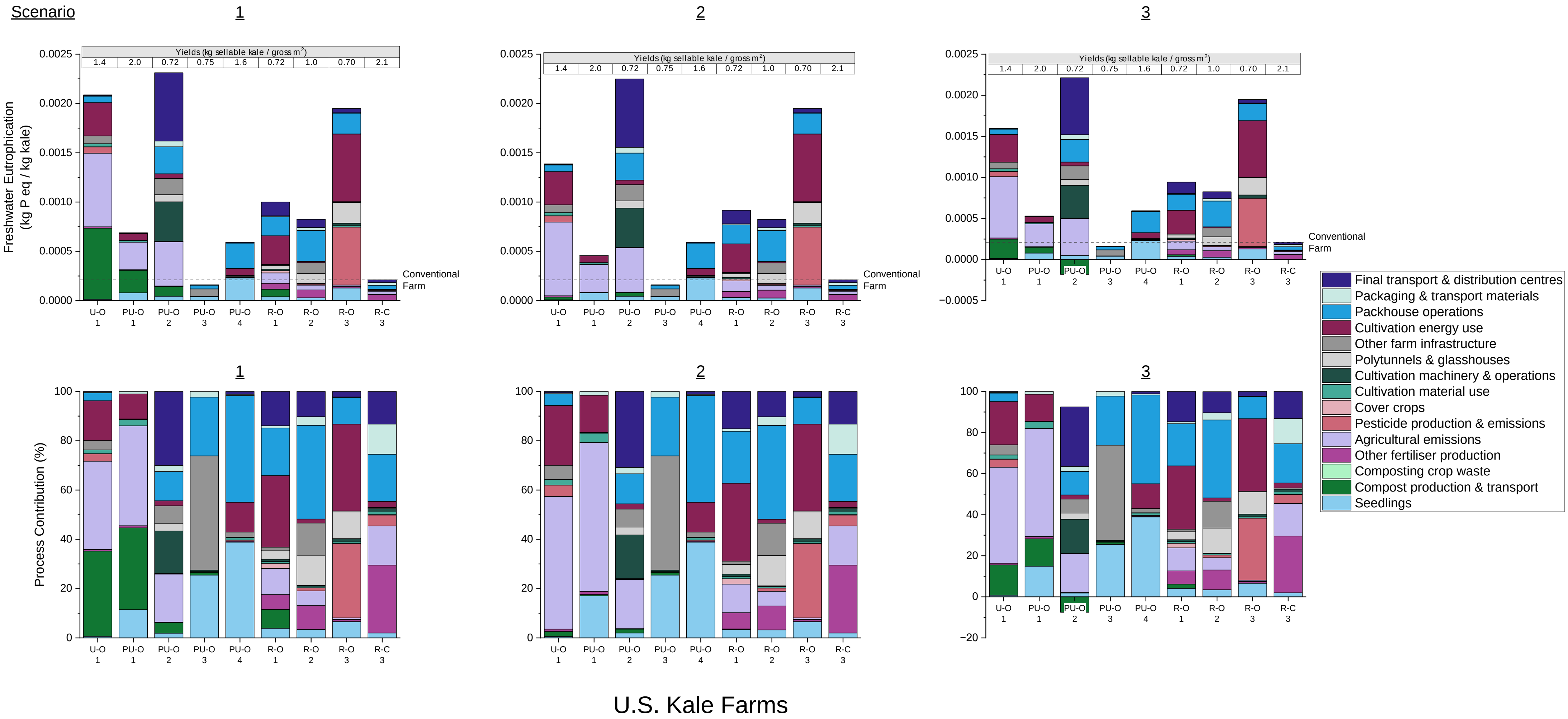

***Figure 27** – Freshwater eutrophication impacts and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg P eq per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

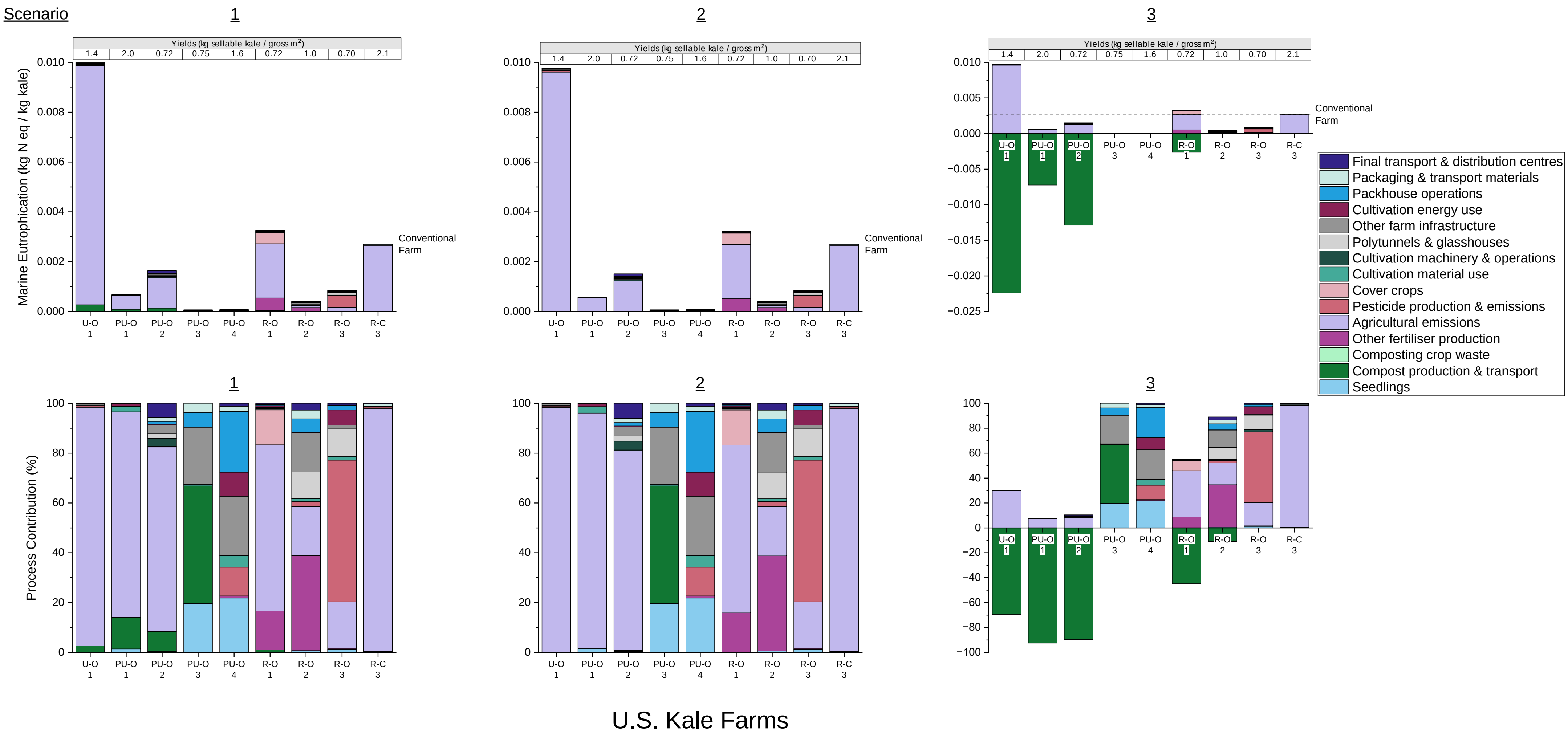

***Figure 28** – Marine eutrophication impacts and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg N eq per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

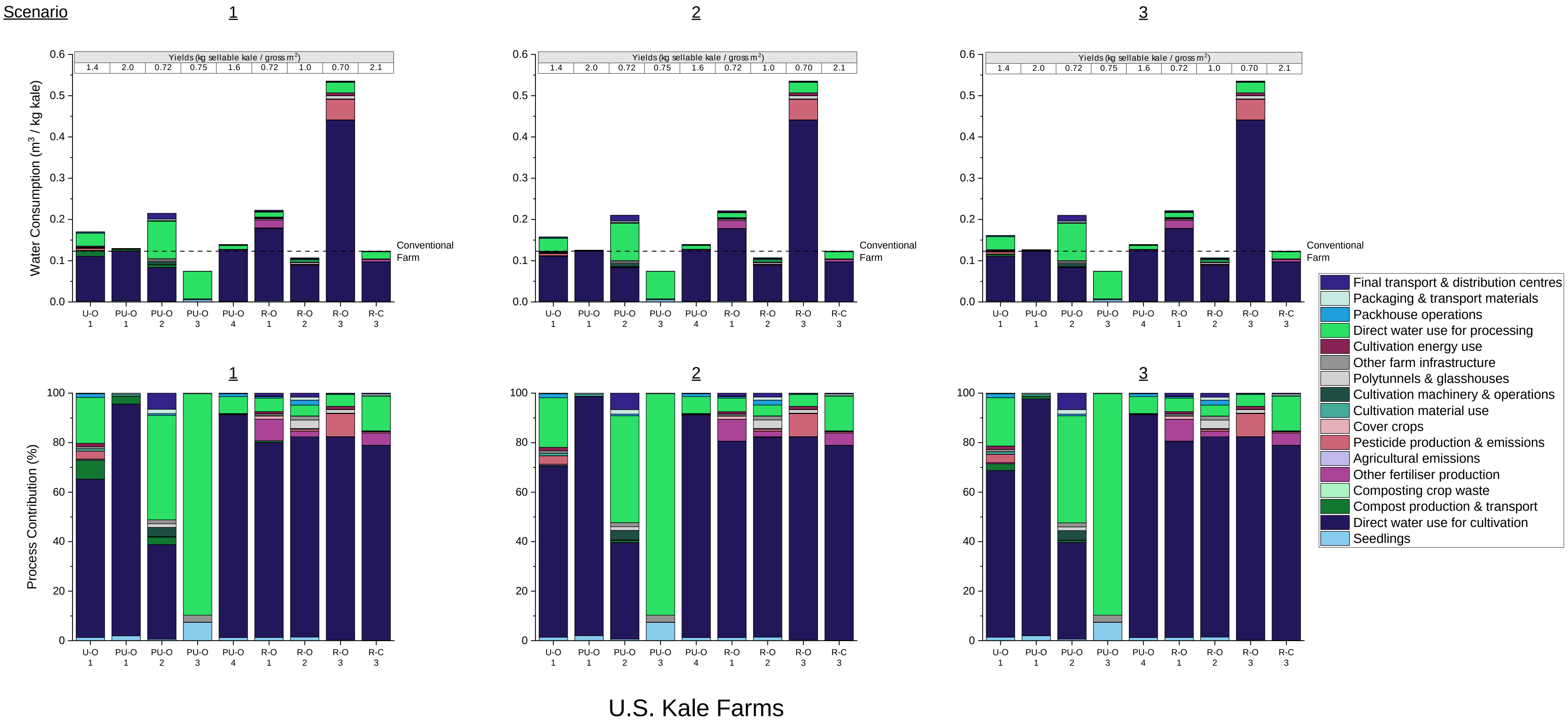

Figure 29 *– Water consumption impacts and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in m^3 per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

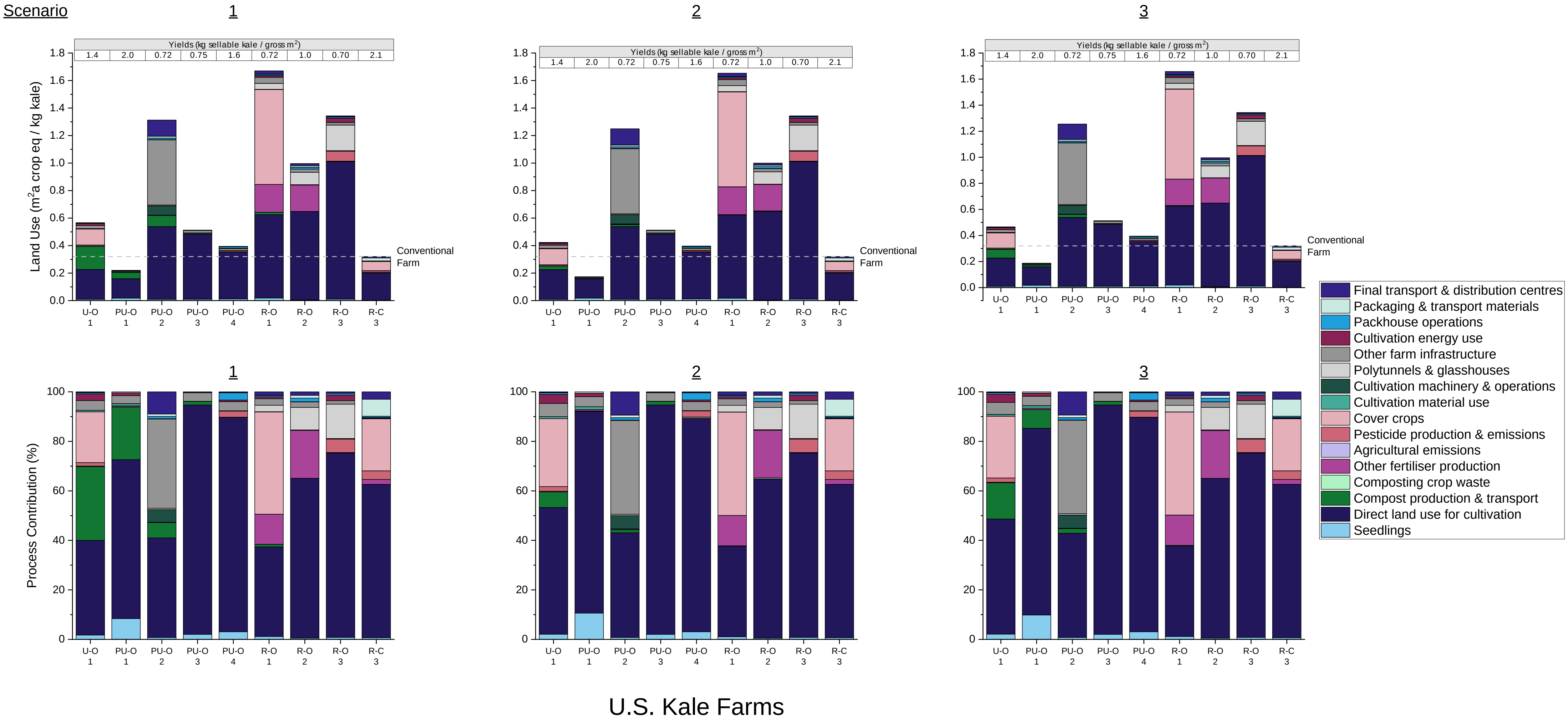

***Figure 30** – Land use impacts and contributing processes for individual U.S. kale farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in m^{2}a crop eq per kg kale transported to the final point of sale, with a dotted line designating the impact level of the conventional farm. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

3.1.2.2 UK Kale

Figure 31 through Figure 37 display respective global warming, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, water consumption, and land use impacts across individual farms for UK kale production. These results are discussed in a similar format as for U.S. kale production.

3.1.2.2.1 Global warming, fine particulate matter formation, and terrestrial acidification

The major contributing processes to global warming potential (Figure 31), fine particulate matter formation (Figure 32), and terrestrial acidification (Figure 33) for UK kale production include: compost and other fertiliser production, agricultural emissions, cultivation machinery and operations, farm infrastructure, and final transport. The contribution from agricultural emissions is especially important for the latter two impact categories.

Kale production on UK organic farms can largely be characterised into two types. PU-O-1, PU-O-2, PU-O-3, and R-O-1 all grow kale on a relatively small, intensive, market garden scale, using compost as the main fertility input. In contrast, U-O-2, PU-O-4, and R-O-2 utilise cover crops as the main fertility input and grow kale on a much larger field scale, requiring the use of tractors. For U-O-2 and R-O-2, cover crops are the sole fertility input, whilst PU-O-4 supplements this with the application of fresh manure. Thus, the main factors contributing to impacts on organic farms will vary between these two groups of farms, based on these two modes of production.

The organic farms using compost as an input tend to have some of the highest impact levels in Scenario 1 (e.g., PU-O-1 and PU-O-3), as also seen for U.S. kale production. However, these farms also tend to have some of the highest yields in comparison to the other organic farms growing kale on a field-scale, likely due to the more intensive production. In some cases, yields are even higher than that achieved on the conventional farms. Impacts for PU-O-1, PU-O-2, PU-O-3, and R-O-1 are thus largely driven by compost production, generally seen to contribute to >50% of impacts. For farms applying the highest amounts of compost (PU-O-1 and PU-O-3), contribution from compost production constitutes nearly 80% of total global warming impact. For fine particulate matter formation and terrestrial acidification, agricultural emissions (from compost and other fertiliser application) are another main impact driver for these farms, contributing to 20-50% of impacts. Thus, for these two impact categories, impacts are driven almost exclusively by compost production and agricultural emissions for these farms.

Although the farms using compost as an input tend to have the highest overall impacts in Scenario 1, particularly seen with PU-O-1 and PU-O-3, some farms using compost were still able to achieve relatively low impact levels. This is seen for PU-O-2 and R-O-1, two farms which applied lower levels of compost by offsetting this with the use of manure. These farms both achieved global warming and fine particulate matter formation impacts lower than the conventional average, although higher for terrestrial acidification. R-O-1 also has one of the highest yields, likely due to their intensive production in polytunnels during a relatively short growing season (5 months vs. average of 8.5 months for other organic farms).

In contrast to the farms using compost and growing kale on a market garden scale, the other organic farms that grew kale on field-scale using cover crops tended to have some of the lowest impacts among all farms. In particular, PU-O-4 and R-O-2 have the lowest overall global warming potential, fine particulate matter formation, and terrestrial acidification impact levels. These two farms are also the only organic farms with terrestrial acidification impact levels lower than the conventional average in Scenario 1. These farms also achieved yields similar to those on conventional farms, although less than those achieved on some of the more intensive organic farms that used compost as an input (e.g., R-O-1 and PU-O-3). The exception, however, is U-O-2, which has relatively high impact levels above the conventional average, albeit lower terrestrial acidification impacts than the farms using compost. This urban farm also had the lowest overall yield (0.26 kg m^{-2}), suggesting that the use of cover crops as the sole fertility input was perhaps not ideal for this farm.

The main impact contributor for U-O-2, PU-O-4, and R-O-2 comes from cultivation machinery and operations, driven mainly by diesel combustion in tractors. This accounts for 45-70% of total global warming potential and terrestrial acidification impacts and 60-80% of fine particulate matter impacts. For U-O-2, the contribution from cultivation machinery and operations (in actual terms) is particularly high compared to PU-O-4 and R-O-2, thus driving the higher impacts observed on this farm. This difference is not because of higher diesel use, but actually because of the higher contributions coming from the production burdens of tractors and implements. This is higher for the urban farm due to its much smaller field scale, growing only a few crops using machinery. Thus, the allocation of tractor and machinery burdens to kale production are higher for this farm than for the other larger-scale farms (PU-O-4 and R-O-2), which use their tractors and machinery across a larger area and for a wider variety of crops.

For the farms using cover crops for fertility, the emissions generated when incorporating cover crop residues into the soil are a minor contributor to impacts. This is because N_2O and nitrate leaching emissions do not contribute to fine particulate matter formation and terrestrial acidification, and for global warming, N_2O emissions from cover crops are largely offset by the CO_2 incorporated within the main crop biomass, which is why negative contributions from agricultural emissions are seen for many organic farms. Impact contributions from cover crops come mainly from the burdens associated with producing cover crop seed, and still this contributes to <30% of global warming impacts and <10% of fine particulate matter formation and terrestrial acidification impacts for farms using cover crops. In real terms, these impacts are always much less than seen for compost production impacts on other farms. This points to the use of cover crops as a potentially more attractive fertility input than compost from an environmental standpoint, although this comes at a slight trade-off with yield. Further, the advantages in terms of global warming potential are largely dissipated in Scenario 2.

For conventional farms, impacts are driven mainly by fertiliser production, agricultural emissions, and cultivation machinery and operations. All together, these comprise approximately 40-60% of global warming potential and 60-80% of fine particulate matter formation and terrestrial acidification impacts. Packaging, farm infrastructure, and final distribution constitute the majority of the rest of impacts seen on conventional farms, with energy use within packhouse operations also a significant contributor for R-C-2. Generally, the conventional farms have higher contributions from packaging materials and packhouse operations than for organic farms, collectively comprising 20-30% of total global warming potential and 10-20% of fine particulate matter and terrestrial acidification impacts, compared to <5% on most organic farms (except PU-O-4, which had a relatively high contribution from packaging). In contrast to U.S. production, impact contributions from final transport (in actual terms) are generally higher on the rural conventional farms in comparison to the other organic farms. This is likely because produce is transported much smaller distances within the UK as opposed to within the U.S. Final transport contributes to approximately 5-12% of impact on the rural conventional farms, with similar contributions (in both actual terms and percent) also seen for PU-O-2, due to the fact that this farms delivers through a city-wide home delivery route.

Differences in the main impact contributors between conventional farms are seen particularly for R-C-1 and R-C-2 vs. R-C-3, R-C-4, and R-C-5. The prior two farms grow kale on a much smaller scale than the others (<2 ha compared to 90-200 ha), with the highest impacts generally driven by cultivation machinery and operations. This accounts for at least a third of total global warming and terrestrial acidification impacts for these two farms, compared to less than 15% on the larger-scale conventional farms (R-C-3, R-C-4, and R-C-5). The larger-scale conventional farms have impacts instead driven more by fertiliser production and application emissions. Although the main contributing processes are distinct between these two groups of conventional farms, impact levels vary; for example, R-C-1 has one of the lowest impacts out of the conventional farms, and R-C-2 one of the highest, indicating that scale does not solely predict impacts.

The two conventional farms with the lowest global warming impacts are R-C-1 and R-C-3. The prior uses the lowest nitrogen input out of all conventional farms (one order of magnitude less than the others), thus decreasing the contribution from fertiliser production and agricultural emissions, which are the major contributing processes to global warming potential on other conventional farms. Even with this low N input, R-C-1 is still able to achieve relatively high yields, perhaps due to the smaller scale of production and the use of irrigation, which was not employed on most other farms. R-C-3, on the other hand, used production practices similar to many of the other conventional farms, but also had the highest yield out of these farms because they were able to grow kale year-round, which lowered overall impacts per kg sellable crop. The highest impacts out of conventional farms are seen on R-C-2, driven largely by a relatively high fuel use for cultivation and also a high energy use for packhouse operations, which is a major impact contributor despite the farm offsetting 20% of energy use with its own solar energy.

In Scenario 2, the variation observed between organic farms using or not using compost are largely dissipated when compost production burdens for compost inputs are not allocated. This results in almost all organic farms having global warming and fine particulate matter formation impact levels lower than or similar to the conventional average in Scenario 2. The exception is U-O-2, which did not use compost, and thus was not affected by the change in scenarios. Additionally, this farm already had relatively high impact levels in Scenario 1 due to its low yield and high impacts from cultivation machinery and operations. For terrestrial acidification, although impact levels for farms using compost are lowered in Scenario 2, still none of these farms achieve impact levels lower than the conventional average. This is because of the much higher contribution from agricultural emissions (spurred largely by compost application) for this impact category, which drives relatively high impacts compared to the other organic farms using cover crops as the fertility input, as the emissions associated with this green manure incorporation are much less. Composting crop waste is also a contributor in Scenario 2, seen to particularly contribute to up to 35% of global warming and terrestrial acidification impacts on some farms, although the contribution of this for farms using compost as an input is much less than the burdens from compost production in Scenario 1.

In Scenario 3, the farms applying the highest amounts of compost now see some of the lowest global warming potentials due to the avoided burdens attributed, as similarly seen for U.S. kale production. In particular, this results in PU-O-1, PU-O-3, and R-O-1 all having overall negative global warming potentials, thus having the lowest impacts for global warming. Negative contributions from avoided burdens in Scenario 3 are not as significant for fine particulate matter formation and terrestrial acidification as for global warming potential. For these impact categories, impacts for farms using compost are slightly lowered in comparison to Scenario 1, but overall trends remain largely unchanged.

3.1.2.2.2 Freshwater eutrophication

In contrast to the previous three impact categories, freshwater eutrophication impacts are driven primarily by agricultural emissions (Figure 34), due to P leaching from compost and fertiliser application. Unlike for U.S. kale production (Figure 27), in the UK energy use is not a major contributor to freshwater eutrophication impacts for kale production. This is due in part to a relatively lower energy use, as irrigation is often not used for kale in the UK and cold storage energy requirements are also minimal due to the UK climate. However, this is also because the UK electricity grid comprises a much smaller proportion of coal power (8% in the ecoinvent process market mix), which is the main driver of freshwater eutrophication impacts from energy use in the U.S. Other drivers of freshwater eutrophication impacts for UK kale production thus stem mainly from cultivation machinery and operations, other fertiliser production, infrastructure, and also packaging for conventional farms.

Again, differences arise between the set of organic farms using compost as an input and those using mainly cover crops for fertility. For freshwater eutrophication, these differences are driven less by compost production burdens and more by the agricultural emissions associated with compost and other fertiliser application. These agricultural emissions account for 45-75% of impacts for farms using compost as an input. Thus, the farms that apply the largest amounts of compost per area (PU-O-1 and PU-O-3) again have the highest overall impact levels. However, all other organic farms are able to achieve impact levels lower than the conventional average due to the more moderate use of compost and other fertilisers and thus lower agricultural emissions.

The lowest impact levels are again observed on the farms using cover crops as the main fertility input. This is because P leaching is assumed to be zero for cover crop incorporation, and thus the use of cover crops do not contribute to agricultural emissions. The main impact driver for these farms is from cultivation machinery and operations, stemming mainly from the burdens associated with the production of machinery rather than fuel combustion, which does not contribute to freshwater eutrophication. This is again why R-O-2 and PU-O-4 have the lowest overall impacts, in comparison to U-O-2 which sees higher impacts from cultivation machinery because of this farm’s small scale (thus having more burdens attributed to kale production). Farm infrastructure is also a significant contributor to burdens for certain organic farms, constituting up to 20% of impacts, especially significant for farms using shipping containers for crop storage due to steel production burdens (e.g., U-O-2, PU-O-2, and PU-O-4).

For conventional farms, the highest contributor to freshwater eutrophication impacts is generally from agricultural emissions due to P fertiliser application, comprising 25-45% of impacts across most conventional farms. The exception is R-C-2, which applied an extremely low level of P fertiliser per area compared to other conventional farms, thus having negligible contribution from agricultural emissions. On the other hand, R-C-4 and R-C-5 show the highest impact levels out of the conventional farms because they applied the highest levels of P fertilisers per area. Other impact contributors for conventional farms include infrastructure, packaging, and final transport. Packaging is seen to be a particularly high contributor (approximately 30% of total impacts) for farms using cardboard box packaging in comparison to packing into plastic trays, specifically R-C-1 and R-C-3. Finally, R-C-2 also has a high contribution from packhouse operations due to energy use, comprising 40% of total impacts.

As freshwater eutrophication impacts are not driven to a significant extent by compost production, no major differences in trends are observed in Scenario 2, where compost production burdens are removed. Additionally, the composting of crop waste does not contribute to impacts in Scenario 2 because the fugitive emissions from compost production do not contribute freshwater eutrophication impacts. Scenario 1 sees contributions from compost production only because of the energy use estimated for industrially-produced composts.

On the other hand, the avoided burdens from the municipal solid waste stream do contribute to negative contributions in Scenario 3, to a much higher extent than seen for U.S. production. This is because of the higher phosphorus and phosphate emissions arising from incineration (constituting 65% of British municipal solid waste management) compared to landfilling (considered to be 100% of municipal solid waste management in Georgia). Thus, the avoided burdens applied to compost production for UK farms result in PU-O-1 and PU-O-3, the farms with the highest impacts in Scenario 1, now also having impact levels lower than the conventional average.

3.1.2.2.3 Marine eutrophication

As for U.S. kale production, marine eutrophication impacts for UK kale farms are driven almost solely by agricultural emissions, constituting >90% of impacts for almost all farms (Figure 35). This is driven mainly by nitrate leaching from compost and fertiliser use and cover crop incorporation. The exception is R-O-1, which did not have any contribution from agricultural emissions. This is because the calculated N flux for this farm was negative, thus resulting in an assumption of zero nitrate leaching (see: Equation 11 for calculation information). Further, this farm was also the only organic farm to grow kale under cover (in polytunnels), and thus precipitation is also not considered for nitrate leaching calculations. For R-O-1, contributions arise mainly from industrial compost production and polytunnel and infrastructure burdens. However, in actual terms the contributions from these are minimal.

High levels of variation are observed between all farms for marine eutrophication impacts, which are not necessarily related to farm categories or prior trends observed for other impact categories. As expected, R-O-1 has the lowest overall impact level, due to the zero contribution from nitrate leaching, which makes this farm's impact level at least two orders of magnitude lower than all other farms. Relatively low impact levels are also seen for R-C-1, PU-O-4, and PU-O-2, in ascending order of impact. This can be attributed to relatively low total N inputs used on these farms in comparison to those on other farms (both organic and conventional).

For farms with higher impacts (e.g., U-O-2, PU-O-1, PU-O-3, R-C-3, and R-C-4), the reasons for this are more complex because they relate to both N input amounts, but also to the specific climatic and soil conditions that are used to calculate N leaching levels for individual farms (see: Table 55). PU-O-1 and PU-O-3 did apply high levels of compost per area, and this led to relatively high levels of nitrate leaching. On the other hand, U-O-2 has the highest marine eutrophication impact level, despite having a much lower N input from cover crop residues only. The leaching impact on this farm is driven higher due to much higher precipitation levels and lower soil clay content. Additionally, although R-O-2 had some of the lowest impacts for previously discussed impact categories, this farm displays impacts higher than the conventional average for marine eutrophication, although within the range of the conventional farms. This is because the total N input on this farm was actually similar to that used on most conventional farms, even though this came only from cover crops.

For conventional farms, R-C-3 is seen to have one of the highest marine eutrophication impacts, despite having the one of the lowest impacts out of the conventional farms for the prior four impact categories. This high marine eutrophication impact is due to having both the highest N inputs per area, and also the highest precipitation levels out of all the UK farms. R-C-4 and R-C-5 apply similar amounts of N inputs, but marine eutrophication impacts for R-C-4 are also much higher due to higher precipitation levels (approximately double that of R-C-5). Thus, impacts of marine eutrophication can be seen to be driven more by specific geographical context, even within a country, than other impact categories. Additionally, the larger variations of soil and precipitation seen between regions in England, versus regions in Georgia (despite having similar areas), means that this geographical driver of variation is seen even moreso for UK farms.

When comparing across scenarios, little change is observed between Scenarios 1 and 2, since compost production is not a major contributor of impacts for marine eutrophication (as also seen for freshwater eutrophication). However, negative contributions are seen in Scenario 3. This is mainly due to the avoided ammonium and nitrogen emissions associated with landfilling (as part of the UK's municipal solid waste management), which is why these negative contributions were also observed for U.S. farms (where landfilling is the sole municipal solid waste process). These negative contributions result in the PU-O-3 and R-O-1 now having overall negative impact levels. This leaves U-O-2 and R-O-2, which do not use compost, as the only organic farms with marine eutrophication impacts higher than the conventional average in Scenario 3.

3.1.2.2.4 Water consumption

Water consumption for UK kale production appears to vary drastically between farms (Figure 36), and this is largely attributed to the fact that many farms do not irrigate kale, instead relying solely on rain. Thus, for farms that do irrigate, water consumption for cultivation purposes is the main contributor to impact, but for other farms, impact contributors are more variable. R-O-1, a farm that was seen to have some of the lowest impact levels for other categories, has the highest water consumption due to having the highest irrigation rate for kale. This farm was the only one to produce kale inside polytunnels, and thus required more irrigation since no precipitation reached the crop. R-C-1 and R-C-2, the conventional farms growing kale on relatively smaller scale, were also the only two conventional farms to irrigate kale, and thus had the highest water consumption levels out of the conventional farms. Impact levels between all other organic and conventional farms are largely similar, even for those organic farms that did use some irrigation. The lowest levels of water consumption are seen for PU-O-3 and PU-O-4, which also did not irrigate their kale, and also for R-O-2, which irrigated kale only slightly when transplanting into the field.

For farms not using irrigation, a variety of other factors contributed to water consumption impacts, including water use for seedlings, compost production, other fertiliser production (especially for liquid fertilisers), cultivation machinery and operations, and also cultivation and packaging materials (mainly from cardboard and plastic use). Direct water use during storage and processing is only seen for conventional farms, as washing kale is not as common of a practice for the UK organic farms as it is in the U.S. Still, this is generally a minor contributor of water consumption for UK conventional farms (<3%), except for R-C-5, where this constitutes approximately 30% of impact. Otherwise, fertiliser production and packaging materials are the main contributors to water consumption for the conventional farms that did not irrigate (R-C-3, R-C-4, and R-C-5).

Unlike for U.S. kale production, there are more differences observed for water consumption levels between scenarios for UK kale production. This is mainly due to the fact that compost production is a more significant contributor of water consumption impacts for organic farms, only because many did not irrigate and thus had few other main contributing processes. Thus, water consumption levels are slightly decreased for farms using compost as an input in Scenario 2, although this does not significantly change results.

In Scenario 3, there are also negative contributions from the avoided burdens applied to compost production, which was not seen as clearly for U.S. production because in the UK, this is driven by the relatively high water consumption associated with incineration as opposed to landfilling. These avoided burdens do not largely affect R-O-1, which has high water consumption impacts from irrigating, and PU-O-2, which applied very low levels of compost, but it does result in PU-O-1 having the second lowest impacts and PU-O-3 having an overall negative water consumption.

3.1.2.2.5 Land use

Land use for UK kale production is driven mainly by direct use for cultivation and for cover crops (Figure 37), as for U.S. kale cultivation. Contributions from cover crops are particularly high for (approximately 50%) for U-O-2 and R-O-2, which grow their cover crops for as long as their kale season and use this as the sole fertility input. For R-O-1, additional contribution to land use comes from farm infrastructure (43% of total impact), mainly because of the wood used in their farm shed.

As expected, land use is highly correlated to crop yield. Thus, in contrast to most other impact categories, the organic farms with the lowest land uses are mainly those that apply compost as a fertiliser because these farms have the highest yields. This includes PU-O-3, R-O-1, and PU-O-1, in ascending order of impacts, with the first two having impact levels lower than the conventional average. Unlike for U.S. kale production, for UK production the conventional farms did not always have higher yields than the organic farms, and thus have land uses within the range of other organic farms. The farms with the highest land uses are the organic farms that use cover crops as the sole fertility input (U-O-2 and R-O-2); this is due to both the additional land contributions from using cover crops, and also because these farms had the lowest yields. This thus presents a trade-off in impacts, as generally these farms had some of the lowest impacts for other categories because of the use of cover crops, but this also requires more land use.

Comparing across scenarios, trends remain largely unchanged because compost production is not a significant contributor to land use in comparison to other direct land uses for cultivation or cover crops. Land use is only slightly reduced for farms using compost in Scenario 2, and even less so in Scenario 3.

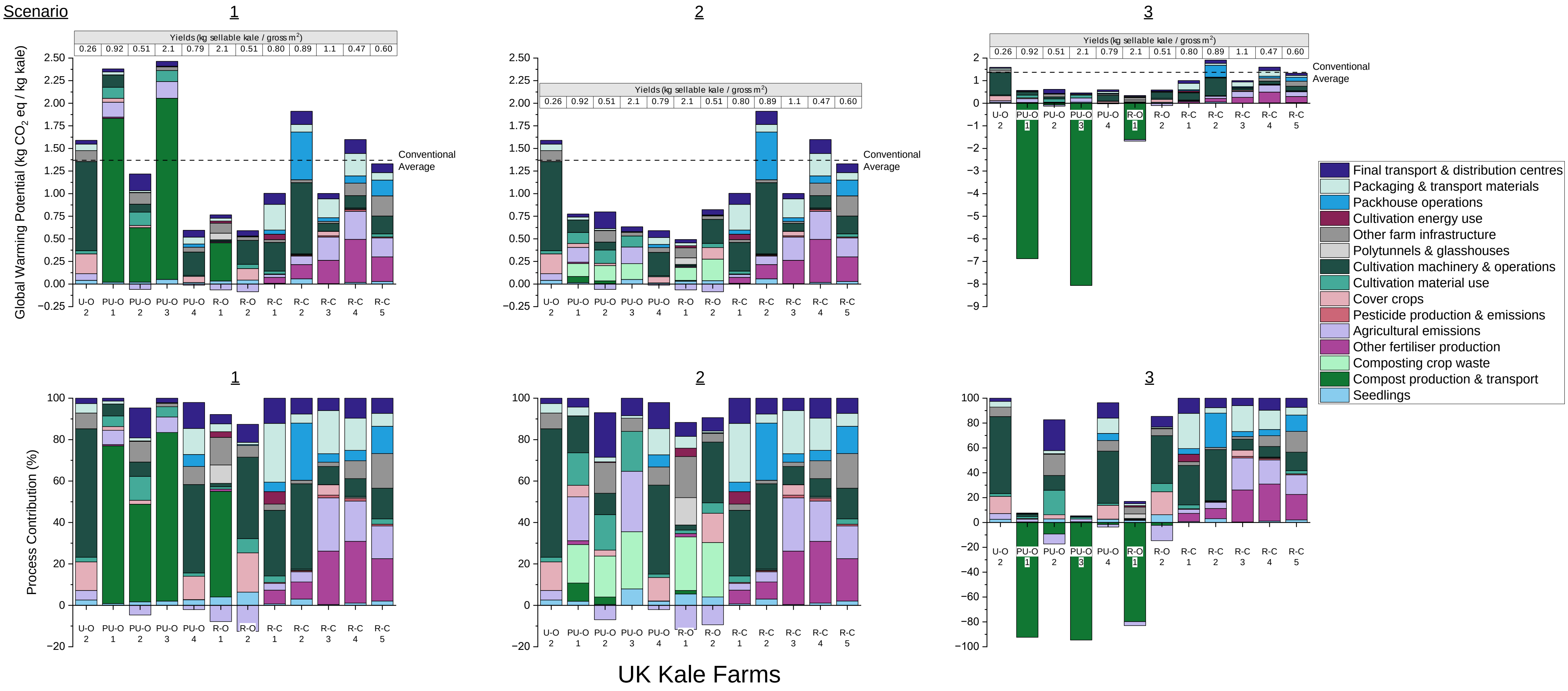

Figure 31 *– Global warming potential and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in kg CO_2 eq per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

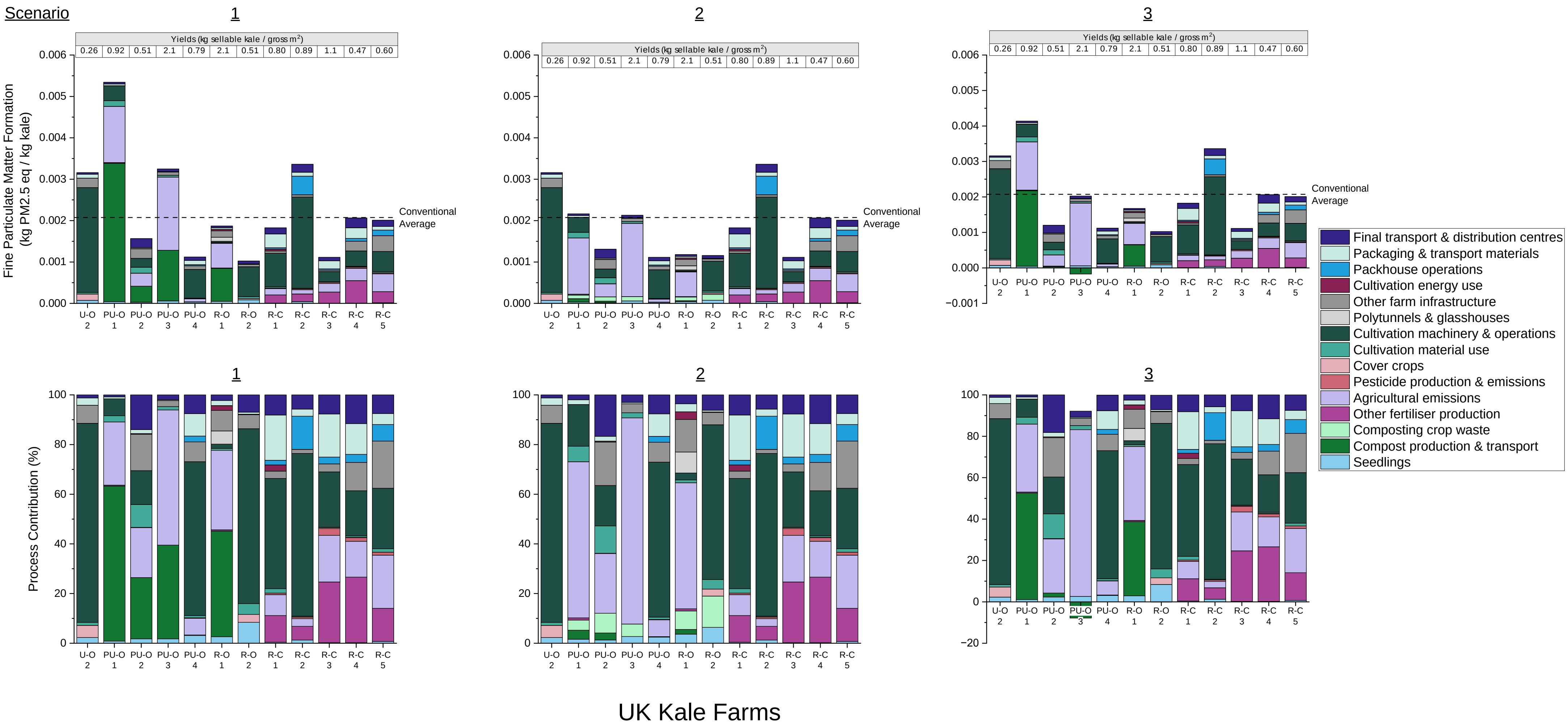

Figure 32 *– Fine particulate matter formation impacts and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in kg PM2.5 eq per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m*$^{-2}$*).*

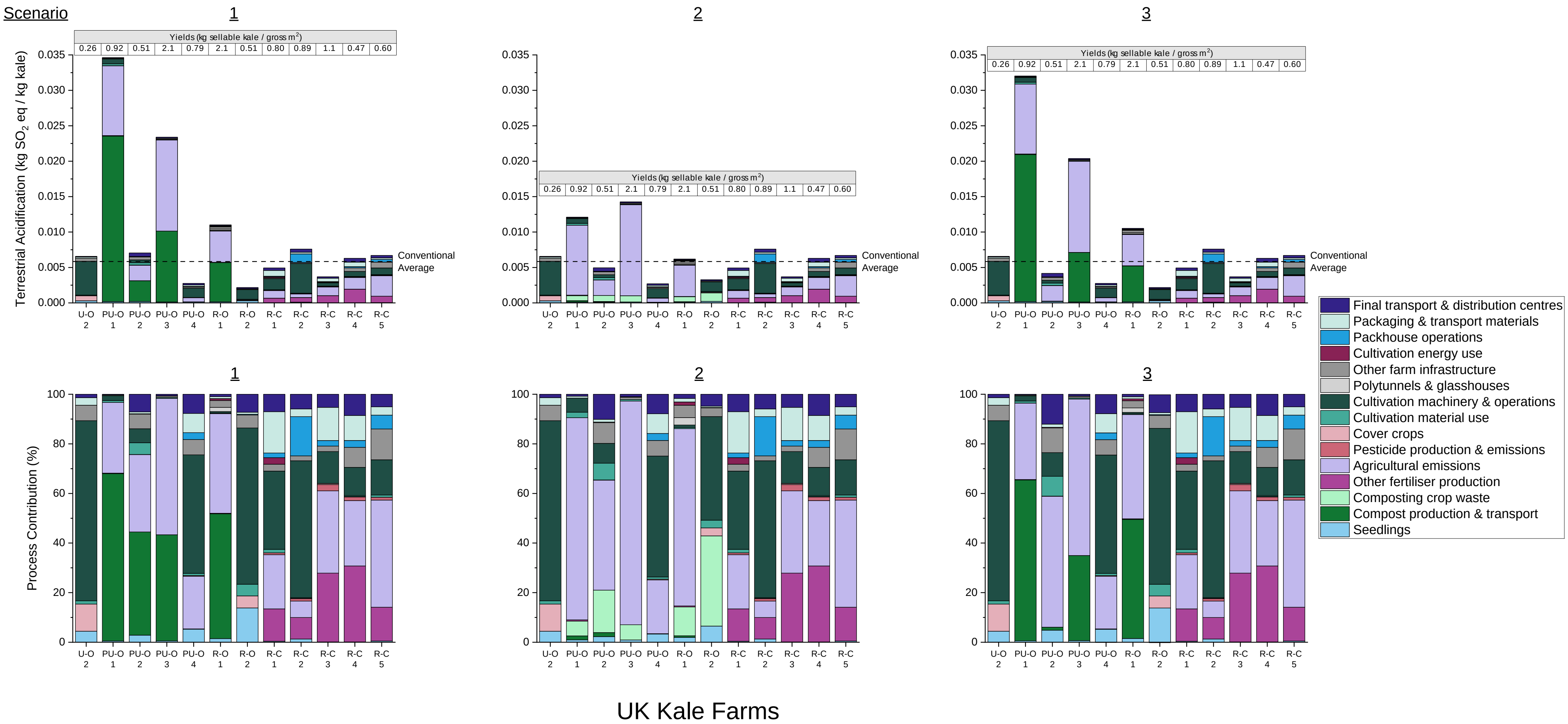

***Figure 33** – Terrestrial acidification impacts and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in kg SO_2 eq per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

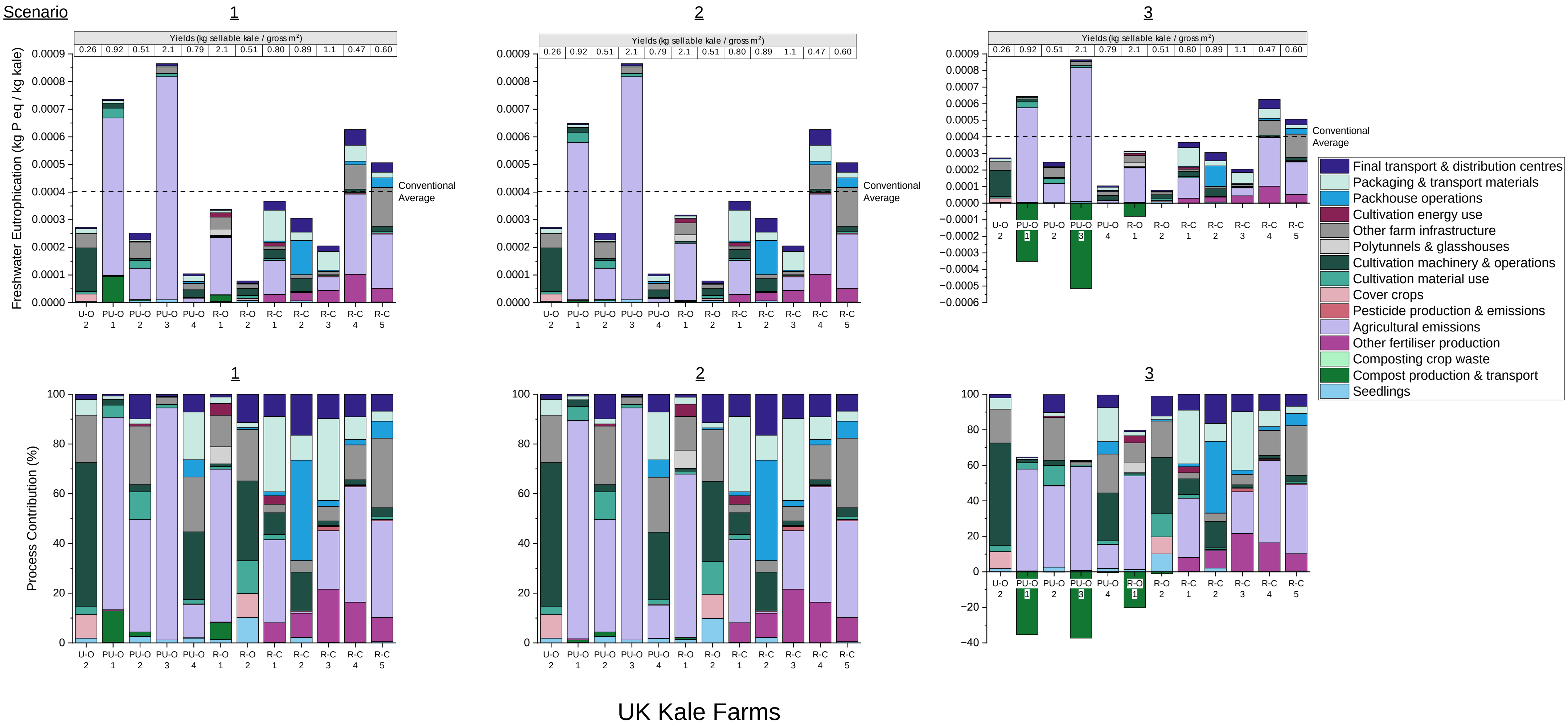

***Figure 34** – Freshwater eutrophication impacts and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in kg P eq per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

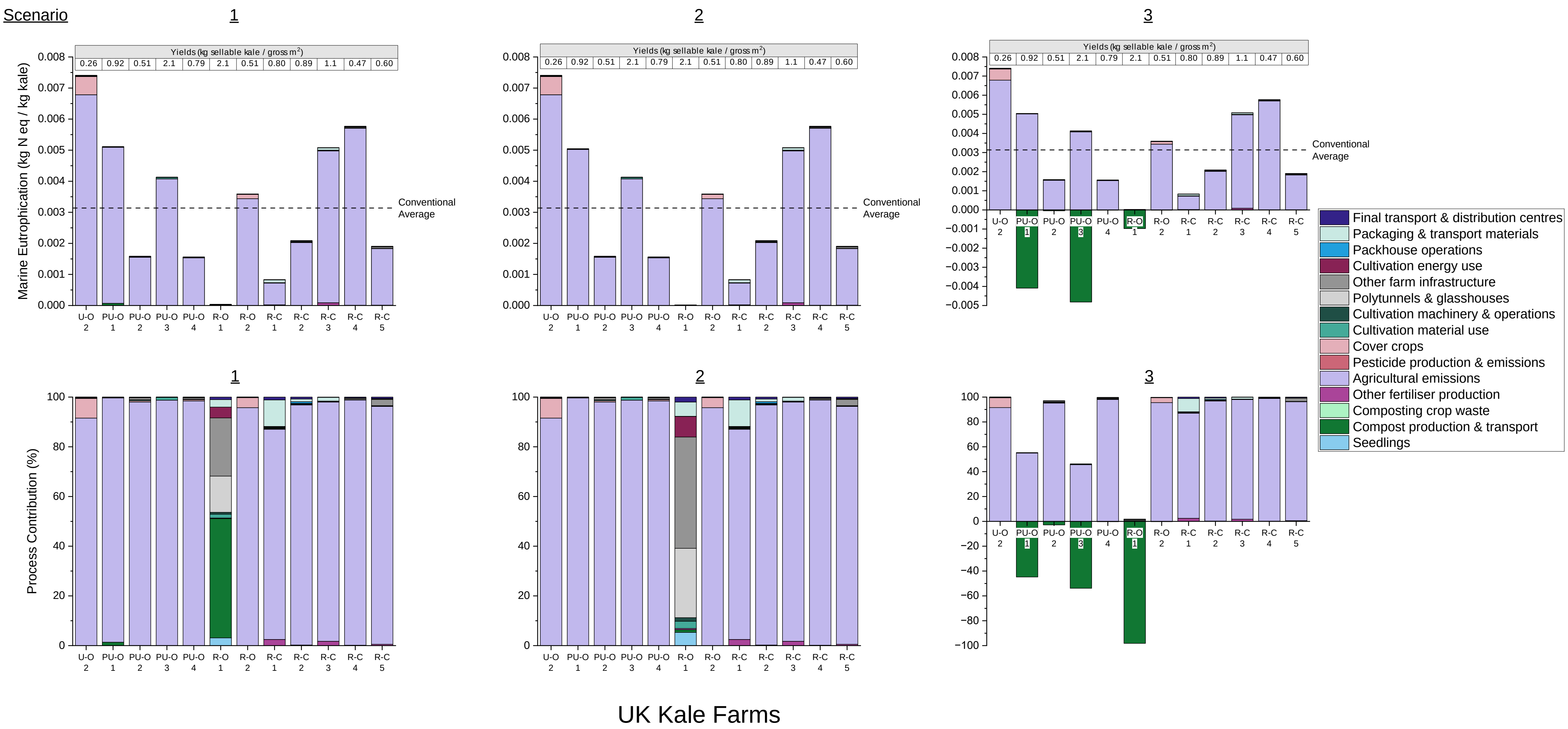

Figure 35 *– Marine eutrophication impacts and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in kg N eq per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

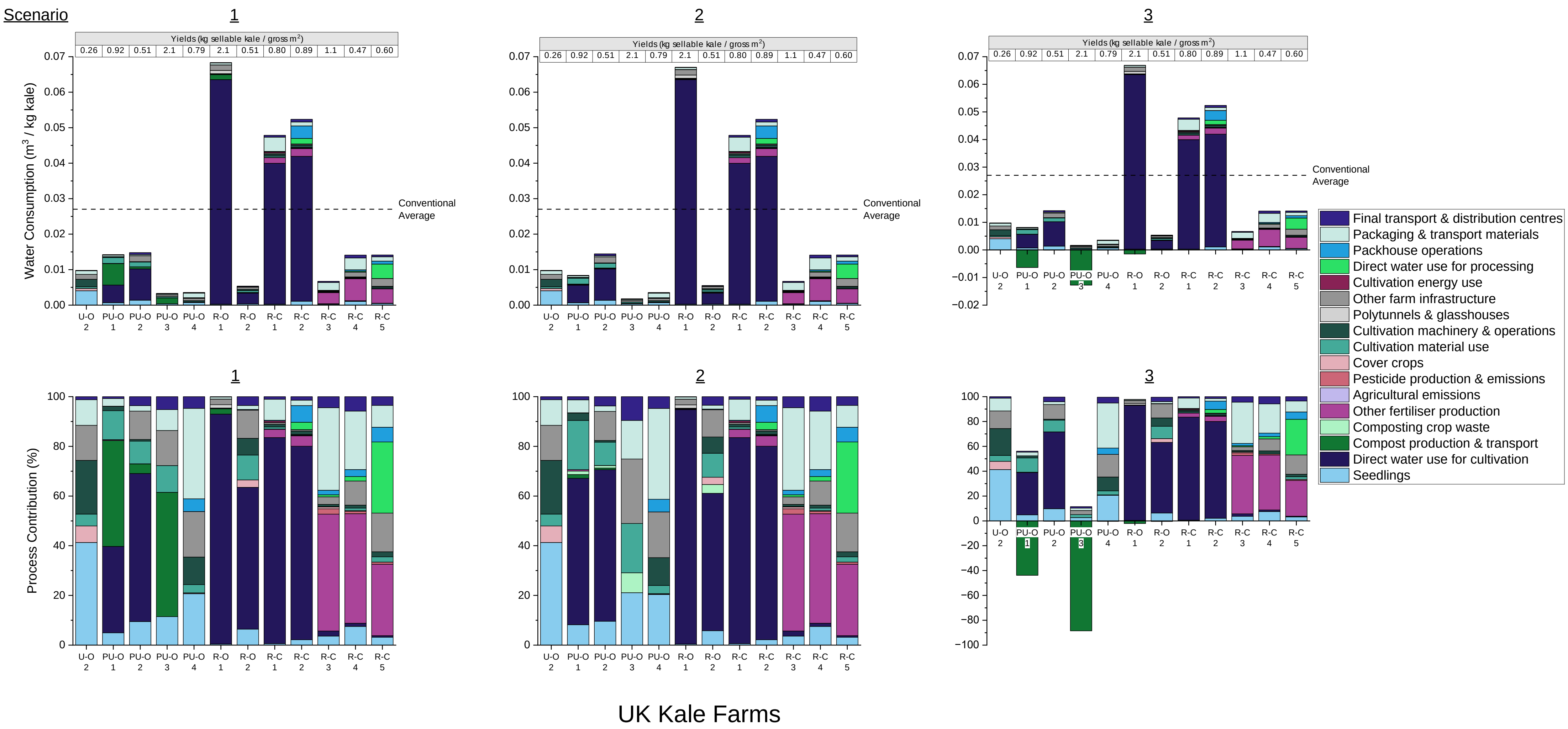

Figure 36 *– Water consumption impacts and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in m^3 per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

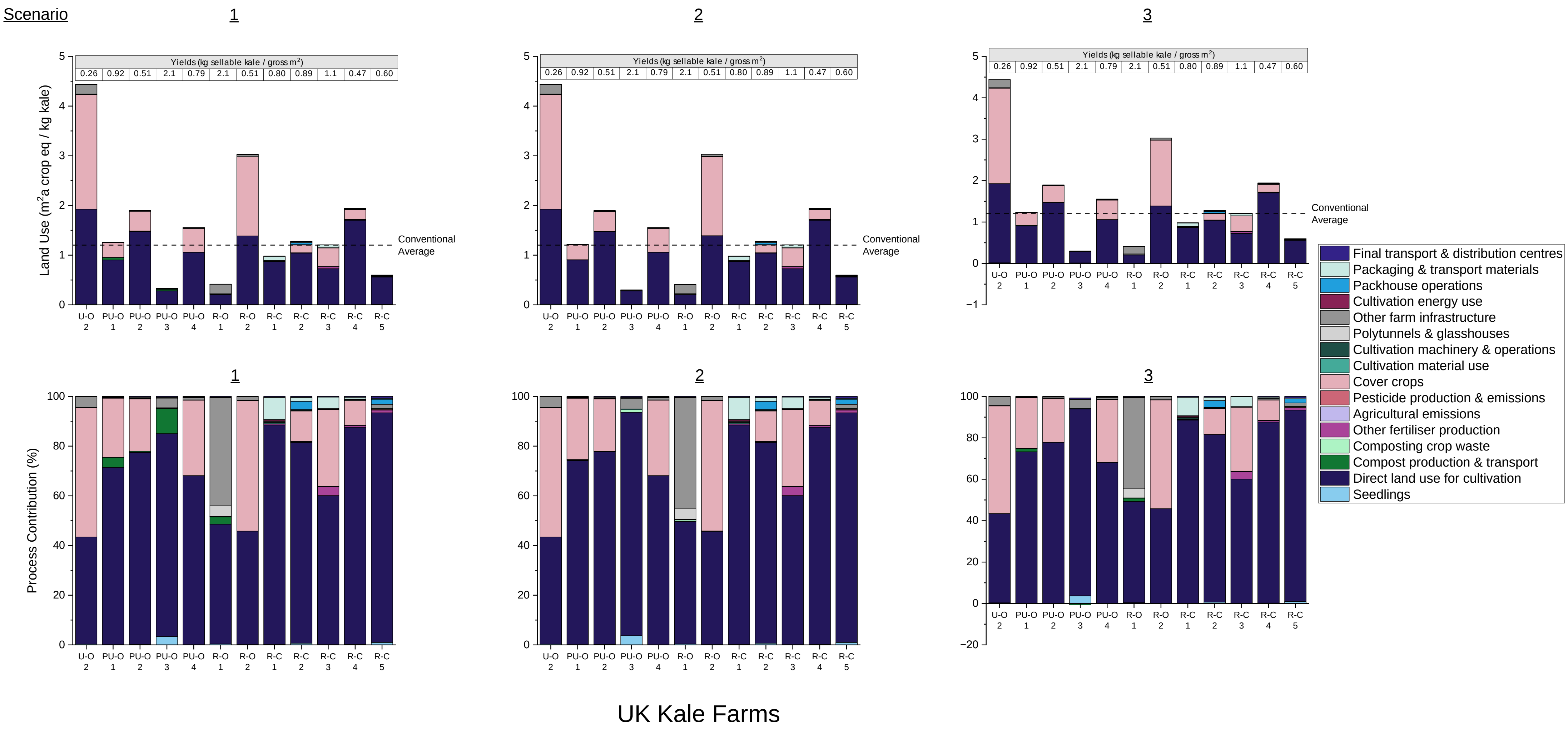

Figure 37 *– Land use impacts and contributing processes for individual UK kale farms (n=12), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. From left to right across the page, each individual plot signifies results from allocation Scenarios 1-3, as numbered. The top row displays actual impacts in m^2a crop eq per kg kale transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable kale (gross m^{-2}).*

3.1.2.3 U.S. Tomatoes

Figure 38 through Figure 44 display respective global warming, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, water consumption, and land use impacts across individual farms for U.S. tomato production. Trends between individual farms, and the main processes contributing to these trends, are discussed within this section.

3.1.2.3.1 Global warming, fine particulate matter formation, and terrestrial acidification

Global warming (Figure 38), fine particulate matter formation (Figure 39), and terrestrial acidification impacts (Figure 40) for U.S. tomato production are influenced by a wide array of factors, which drive the high variation observed between organic farms. These differences are not particularly tied to farm category, suggesting that individual management practices, moreso than farm scale or rurality, are driving the differences in impacts. On the other hand, impact levels for the conventional farms (R-C-1 and R-C-2) are more similar due to their similar cultivation, processing, packaging, and distribution methods.

Although almost all organic farms utilise compost as a fertility input for tomato production (except R-O-2), compost production is not necessarily the main driver of impacts for these farms in Scenario 1, unlike for U.S. kale production. This is because of the higher resource and material use required for tomato cultivation versus kale in general (e.g., higher fertility requirements, trellising, etc), which means there are more processes contributing to overall impacts. For the farms that apply the highest amounts of compost (U-O-1, PU-O-2, and R-O-1), compost production is a major contributor of impacts, especially for terrestrial acidification. These farms also have some of the highest overall impacts, driven in part by this compost production burden. For the urban farm in particular, compost production accounts for the majority of impacts, constituting approximately 70% of global warming and fine particulate matter formation impacts and 85% of terrestrial acidification impacts. This is because this farm applied much higher levels of compost than any other farm (28 kg per cropped m^2 compared to <10 kg m^{-2} on other farms) and used relatively low amounts of other fertiliser inputs. However, for other high impacting farms (e.g., R-O-1, PU-O-2, and R-O-3), compost production contributes less to overall impacts, and the fact that these farms still exhibit relatively high impacts in Scenario 2 further suggests the importance of other contributing factors.

Besides compost production, the main impact contributors on organic farms include cultivation materials (mainly from trellising), polytunnels and other farm infrastructure, other fertiliser production, agricultural emissions (particularly for fine particulate matter formation and terrestrial acidification), cultivation energy use, and final transport. Higher contribution from cultivation material use is seen for PU-O-1, PU-O-3, and R-O-3. This is because these farms, in contrast to the other organic farms, grew a significant portion of tomatoes outside of polytunnels and thus had to use steel posts or cages for trellising, items which are often used for only one or two crops a year. The emissions associated with steel production for these materials thus drive higher impacts. On the other hand, the organic farms that grow more (or all) tomatoes indoors generally have higher contributions from polytunnel materials, as seen for U-O-1, PU-O-2, R-O-2, and R-O-3. Cultivation energy use is also a significant contributor for some farms (e.g., PU-O-1 and R-O-3), but generally energy use within packhouse operations is not a major contributor like it was for kale (Figure 24-Figure 26, because tomatoes do not require cold storage. In addition, all three rural organic farms have relatively high contributions (in both real terms and percent) from final transport, as also seen on these farms for kale production. This is due to the longer distances that tomatoes must travel from these farms in comparison to the urban and peri-urban farms, and the relatively inefficient modes of transport that are used in comparison to larger-scale conventional farms (i.e., transport in petrol vans that are not fully packed).

The farms with the highest global warming, fine particulate matter formation, and terrestrial acidification impacts are R-O-1, R-O-3, and PU-O-2, in descending order. The factors driving impacts on these farms are specific to particular management decisions made on each farm. Impacts for R-O-1 are mainly a result of the high level of inputs used for cultivation, including compost, peat moss for indoor beds, and a wide array of other fertilisers; all together, the production of these inputs contributes to approximately 25% of total global warming potential for this farm. Fine particulate matter formation and terrestrial acidification impacts are also driven by compost and other fertiliser production, as well as the agricultural emissions associated with their application, all together comprising 40-50% of these impacts. When compost production burdens for compost inputs are removed in Scenario 2, the impact levels for R-O-1 do decrease and are more similar to R-O-3, but are still the highest, highlighting the importance of other factors in driving this farm's impacts. This farm was also seen to have a relatively low yield (1.3 kg m^{-2}) , approximately half that of the other rural organic farms, despite the high levels of compost and other fertilisers applied. This low yield is associated with this farm's high waste level (30% total), which was related to pest and quality issues. Finally, this farm also sees an exceptionally high contribution from seedling production compared to other farms, which constitutes approximately 20-30% of total impacts for all three impact categories. This is because of compost use for starting seedlings as well as the electricity use for heating them.

Energy use is also a main impact contributor for R-O-3, which is seen to have relatively high impacts despite having an above-average yield (2.7 kg m^{-2} compared to the organic average of 2.3 kg m^{-2}). R-O-3 was the only farm to use heating for tomato crops during greenhouse cultivation, which especially contributed to global warming potential impacts (approximately 30%). Although this practice did extend the growing season for this farm, important when selling directly to consumers, it did not result in high enough yields to offset the associated environmental burdens. Considering that R-O-1 and R-O-3 are the farms with the highest impact levels, it can be seen that these farms are driving the relatively high averages for the rural organic farm category in U.S. tomato production (Figure 17-Figure 19). The other rural organic farm (R-O-2), however, has some of the lowest impacts out of all organic farms, again highlighting the importance of individual management practices in driving impacts over simple designations of farm scale or rurality. Finally, PU-O-2 is seen to have relatively high impacts for tomato cultivation as it also did for kale, although not emerging as a significant outlier in this case. High impacts for this farm are again driven by its inefficient final transport, as well as compost production and polytunnel use.

Unlike for U.S. kale production (Figure 24-Figure 26), for tomato production no organic farms have global warming, fine particulate matter formation, or terrestrial acidification impact levels lower than the conventional farms in Scenario 1 or even Scenario 2. Only one peri-urban farm, PU-O-1, has impact levels comparable to the conventional levels for all three impact categories. This farm also had a relatively high yield (3.7 kg m^{-2}, similar to conventional yields) and was able to achieve this with minimal compost use by supplementing with manure produced on the farm. Additionally, this farm sold all produce through a CSA on the farm site and thus had no transport burdens, which is likely another reason for its lower impacts compared to other peri-urban and rural organic farms. R-O-2 is another farm seen to have relatively low impact levels, especially for terrestrial acidification, where levels are similar to that of the conventional farms. These low impacts are attributed mainly to the relatively high yield (2.7 kg m^{-2}) and low fertiliser inputs used, as this farmer produced compost tea from relatively low amounts of worm castings and seaweed, thus maximising nutrient benefits. Finally, although PU-O-3 had some of the lowest impacts for kale production, this was not seen for tomato production on this farm. This farm did have fairly low compost and fertiliser inputs per area, as for kale production, but the extremely low tomato yield (0.39 kg m^{-2}) on this farm drove higher impacts per kg of crop. This low yield may be attributed in part to issues with soil fertility and compaction as identified by the farmer, but also likely because this farm did not use any irrigation for tomatoes, unlike all other farms.

For the conventional farms, which had the lowest overall impacts, the main impact drivers are largely similar as to that seen for conventional U.S. kale production. A higher contribution is seen from cultivation materials, however, because of the plastic mulch and trellising materials used for tomato production (approximately 6-10% of total impacts). Additionally, for tomato production there is no contribution from packhouse energy as there was for kale, because tomatoes are generally not stored but packed directly into lorries for transport in the U.S. Fertiliser production is one of the highest contributors, constituting approximately one quarter of the global warming impacts. Additional contributions from the agricultural emissions associated with the use of these fertilisers are particularly important for the other two impact categories; collectively, fertiliser production and agricultural emissions constitute approximately 35% of fine particulate matter formation impacts and 50% of terrestrial acidification impacts. Additional contributions are seen from pesticide production and packaging materials (approximately 7-17% of impacts each), which are not seen for organic farms, although these contributions are low in real terms. Finally, the rural conventional farms do see a significant portion of impacts from final transport (approximately 25% for global warming and fine particulate matter formation and 15% for terrestrial acidification), but in real terms, the contribution is much lower than that seen for the rural organic farms.

Comparing across scenarios, it can be seen that Scenario 2 does lower overall impacts for almost every organic farm, but not enough for any organic farm to achieve impacts lower than the conventional farm average. However, the impact levels for U-O-1, which actually had the highest yield out of all organic farms, do decrease to be more in line with other low-impacting organic farms (e.g., PU-O-1 and R-O-2). This is because of the high contribution from compost production to overall impact for this farm in Scenario 1.

In Scenario 3, the negative contributions from avoided burdens drastically change global warming potential results, as the farms applying the highest amounts of compost are now seen to have the lowest impact levels. This is seen particularly for U-O-1, PU-O-2, and R-O-1, which all had some of the highest global warming impacts in Scenario 1 but have overall negative global warming potentials in Scenario 3. However, negative contributions from avoided burdens only slightly reduce fine particulate matter formation and terrestrial acidification impact levels, and thus the results for these impact categories remain unchanged from Scenario 1.

3.1.2.3.2 Freshwater eutrophication

Freshwater eutrophication impacts for U.S. tomato production are driven by similar processes as for the three previously-discussed impact categories, but to a much larger extent by agricultural emissions, due to associated P leaching from compost and fertiliser application (Figure 41). Additionally, direct energy use and any processes that require high energy use (e.g., production of steel for trellising or polytunnels) are significant contributors, as seen for U.S. kale production. Again, this is because of the phosphorus and phosphate emissions associated with the use of coal power in the southeast U.S. electricity grid. Contributions from compost production are minor for all farms except the urban farm, which applied the highest levels of compost, with burdens associated mainly with energy use during the production of industrially-produced composts.

Trends between farms for freshwater eutrophication impacts remain largely unchanged from the previous three impact categories. R-O-1 and R-O-3 again have the highest impact levels, this time spurred to a higher degree by energy use for cultivation and within packhouse operations, which together comprise approximately 20% of total impacts for these farms. A similar contribution is also seen from seedling production for these farms, due to the energy use for heating during propagation. PU-O-1, PU-O-3, and R-O-3 see relatively high contributions from cultivation materials compared to other farms (12-30%), associated with the energy use for the production of steel used in outdoor trellising materials; similarly, the energy use for steel production in polytunnels drives the contributions from this category for farms growing a majority of tomatoes indoors (e.g., U-O-1, PU-O-2, and R-O-2). Finally, R-O-3 also sees a relatively high contribution from pesticide production (16%), as it did for kale production, due to the emissions associated with the production of insecticidal soap and wastewater generated during this process.

As for the previous three impact categories, no organic farms achieve impact levels lower than the conventional farm average. The organic farms with the lowest impacts (PU-O-1 and R-O-2) are also those applying some of the lowest amounts of compost and other fertiliser inputs per area, thus seeing a lower contribution (in real terms) from agricultural emissions as opposed to other farms. PU-O-1 again has impact levels similar to the conventional farms, with the highest contributions to freshwater eutrophication coming from fertiliser production, cultivation materials, agricultural emissions, and cultivation energy use. For R-O-2, additional contributions are seen from polytunnel infrastructure and also final transport, the latter of which constitutes over a quarter of total impacts. For conventional farms, impacts are driven mainly by agricultural emissions, constituting 50-60% of total impacts. Additional contributions are also seen from the energy use associated with the production of fertilisers and packaging materials, which each constitute approximately 10-15% of impacts.

When comparing across scenarios, no major changes in results are seen because compost production is generally not a major driver of freshwater eutrophication impacts. Additionally, the avoided impacts applied in Scenario 3 result in only slight negative contributions, because the avoided process of landfilling does not contribute significantly to freshwater eutrophication impacts. Thus, while impacts are slightly lowered for the farms using compost in both Scenario 2 and 3, this does not change overall trends as observed in Scenario 1.

3.1.2.3.3 Marine eutrophication

As for other crop lifecycles, marine eutrophication impacts for U.S. tomato production are driven almost solely by nitrate leaching, as accounted for within the agricultural emissions process (Figure 42). This is seen as agricultural emissions account for 75-95% of impacts for the majority of farms, with the exception of only the rural organic farms. Thus, impacts on all other farms are driven primarily by calculated nitrate leaching amounts, which relate to amounts of total N applied as well as climatic conditions. In particular, both conventional farms and PU-O-3 were the only farms to grow tomatoes solely outdoors, in contrast to all other organic farms, which grew at least a portion of tomatoes in polytunnels. Thus, higher amounts of precipitation will be considered for farms growing solely outdoors, which contributes to leaching (see: Equation 11 for calculations).

Agricultural emissions contribute less to impacts for rural organic farms. For R-O-1 and R-O-2, nitrate leaching amounts are assumed to be zero following IPCC (2019) guidelines, as these farms grow tomatoes solely indoors using drip irrigation. For R-O-3, nitrate leaching amounts are lower because this farm cultivates the majority of tomatoes inside using drip irrigation; thus, nitrate leaching is only assumed for the portion of tomatoes grown outside (approximately 20% of total production by area), and thus agricultural emissions contribute to only 20% of impacts for this farm.

The trends observed when comparing impact levels between farms for the four previously-discussed impact categories change when considering marine eutrophication. In this case, several farms have impacts lower than the conventional average, as was also seen for kale production. These include PU-O-1, R-O-2, and R-O-3, the last of which actually has some of the highest impacts for other impact categories. The low impacts seen on these farms in comparison to the conventional farms is due to the lower levels of N inputs and the relatively high yields achieved on these farms, thus resulting in lower amounts of nitrate leaching per crop produced. The lowest impact is seen for R-O-2 because of its estimated zero nitrate leaching.

The farms with the highest marine eutrophication impacts are PU-O-3 and U-O-1. Although the prior actually applied the lowest N input per area out of all farms, this farm actually has the highest marine eutrophication impacts. This is because of the exceptionally low yield seen on this farm (0.39 kg m^{-2}), thus driving higher nitrate leaching amounts per kg. PU-O-3 was also the only organic farm to grow tomatoes solely outdoors, and thus the precipitation amounts that contribute to leaching are higher for this farm. This farm also sees a significant contribution to marine eutrophication from other fertiliser production (15%), which is related to emissions from the cultivation of the hay that this farm buys in to use for compost and mulching.

For U-O-1, impacts are driven almost exclusively by agricultural emissions due to the exceptionally high amounts of compost applied on this farm per unit area, seen even though this farm had the highest yield out of all organic farms (4.0 kg m^{-2}). Finally, R-O-1 was also seen to have a relatively high marine eutrophication impact despite the fact that this farm was assumed to have zero nitrate leaching. For this farm, marine eutrophication impacts are driven primarily by fertiliser production (80% of impacts), which is particularly associated with this farm’s use of seed meals as fertilisers; thus, contributions to nitrate leaching from the production of the plants grown to make seed meals drive impacts. It should be noted that this process presents some degree of uncertainty, as seed meals are assumed to be a co-product of oil production within this LCA because they are being purchased by farms as fertilisers; however, some might argue that this is a by-product that should not be allocated upstream burdens.

When comparing across scenarios, no major differences are observed from Scenario 1 to 2, as compost production does not play a major role in marine eutrophication impacts. However, in Scenario 3, large negative contributions from the avoided ammonium and nitrogen emissions associated with landfilling are seen for the farms applying the highest levels of compost. In particular, this results in U-O-1, PU-O-2, R-O-1, and R-O-3 all having overall negative marine eutrophication impacts. As for kale production, avoided burdens are not applied to PU-O-3, because this farm's compost components would not have alternatively gone to the municipal waste stream (hay and manure). Thus, in Scenario 3 this is the only farm with an impact level higher than the conventional average.

3.1.2.3.4 Water consumption

Water consumption is driven primarily by water use during cultivation (mainly for irrigation), constituting 70-99% of total impacts for most farms (Figure 43). Thus, water consumption levels per kg tomato are driven mainly by irrigation rates, as well as yields. The exception is PU-O-3, the only farm that did not irrigate their tomato crop. For this farm, all water consumption comes from that used for washing tomatoes (processing water use), a practice that was not performed on any other farm. R-O-1 also sees an additional contribution of water consumption from fertiliser production, again associated with the cultivation of plants used to make seed meal fertilisers.

Generally, water consumption increases with increasing rurality for organic farms, as also seen in the averaged farm category results (Figure 22). Thus, the lowest water consumption levels are generally seen for the urban and peri-urban farms, with the highest seen for the rural organic farms, with the exception of R-O-2. In particular, U-O-1, PU-O-1, and R-O-2 have some of the lowest water consumptions overall and also some of the highest yields out of the organic farms. PU-O-3 also has a relatively low water consumption, but also a low yield, due to not irrigating their tomato crop.

The conventional farms fall outside this trend of increasing water consumption with increasing scale and rurality, shown by their relatively low water consumption levels per kg tomato. This is mainly driven by the relatively high yields achieved by these farms. However, differences are also observed between the conventional farms. R-C-1 has a water consumption level over 3x that of R-C-2, mainly due to the fact that this farm resides in a region where precipitation levels are almost half that on R-C-2, and thus this farm requires more irrigation (see: Table 54 for precipitation levels).

Comparing between scenarios, results remain unchanged as compost production is not a major contributor to this category, and the avoided burdens applied in Scenario 3 result in only slight reductions of impact.

3.1.2.3.5 Land use

As for other crop lifecycles, land use for U.S. tomato production is driven primarily by direct land use for crop cultivation, which is related to yield (Figure 44). Additional contribution comes from the land use associated with growing cover crops, seen especially for U-O-1 and R-C-2, as well as from fertiliser production. Fertiliser production is seen to be a major contributor for farms that use plant-based materials as fertilisers or mulches because of the land use impacts associated with the cultivation of these plants (e.g., for hay used by PU-O-1, seed meal used by R-O-1, and alfalfa meal used by PU-O-1 and R-O-2). High contributions are also seen for structures or materials made primarily out of wood, which has high land use impacts from tree production. This is seen in particular for the crop storage infrastructure used by PU-O-2, which is mainly wooden household garages, constituting 40% of total land use impact for this farm. Also, the conventional farms have a relatively high land use contribution from cultivation materials (20%) due to the wooden stakes used for outdoor trellising on these farms.

Because of the additional contributions to land use from cover crops, cultivation materials, and packaging materials seen for the conventional farms, they do not have the absolute lowest land use, despite having the some of the highest yields. The lowest overall land use is actually achieved by PU-O-1, which had the second highest yield out of all organic farms and low contributions from indirect land uses. The conventional farms, with some of the highest yields, then have the next lowest land use impacts. This is followed by U-O-1, which has similar direct land uses to PU-O-1 and R-C-1 because of its slightly higher yields (4.0 kg m^{-2}), but has an overall higher land use due to contributions from cover crops and compost production, each which contribute to approximately 20% of impacts. On the other hand, the farms with the highest land uses have the overall lowest yields, particularly seen for PU-O-3, R-O-1, and PU-O-2, while also having impact contributions from fertiliser production and infrastructure, as previously discussed.

Comparing across scenarios, trends remain the same because compost production is not a significant contributor to land use for most farms, with the exception of U-O-1. Land use is slightly reduced for this farm and other farms using compost in Scenario 2, and is reduced even less so in Scenario 3, thus not changing overall results.

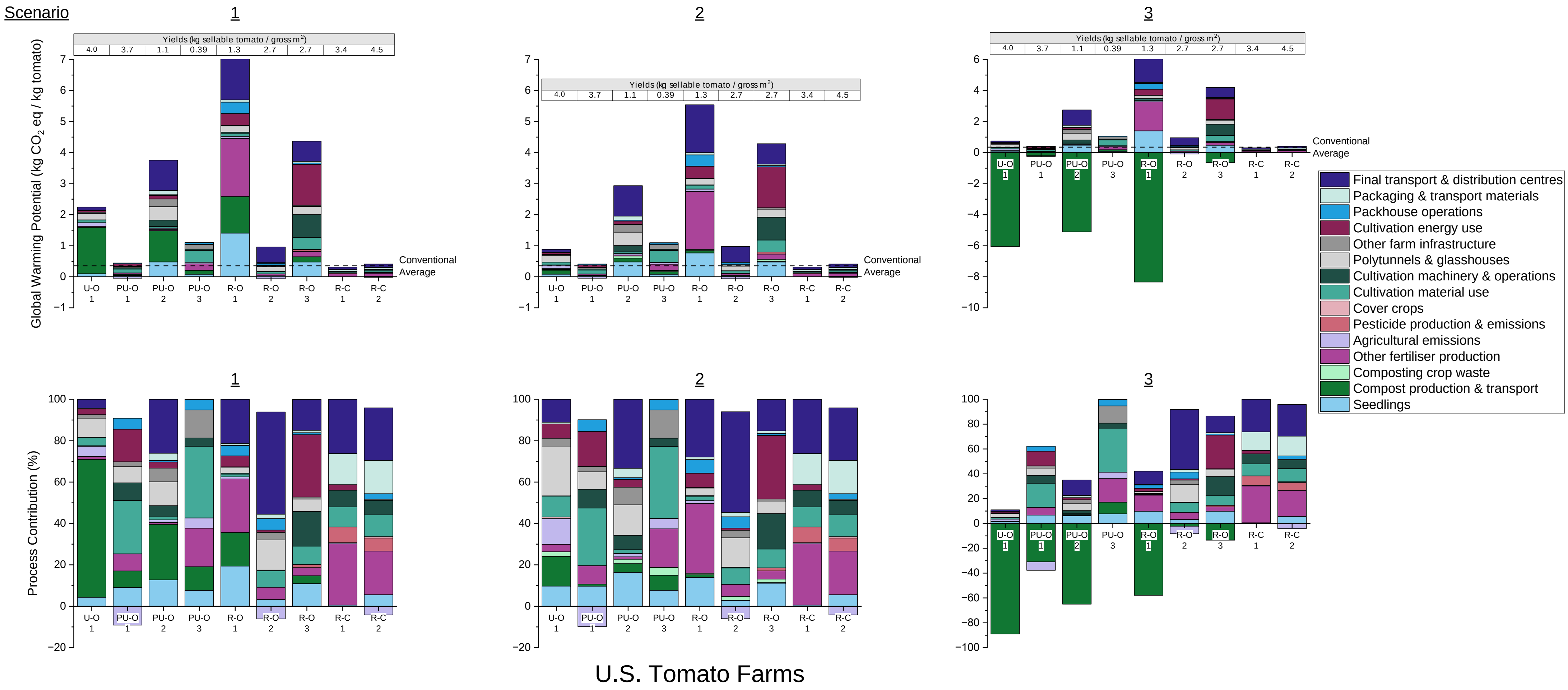

Figure 38 *– Global warming potential and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg CO_2 eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

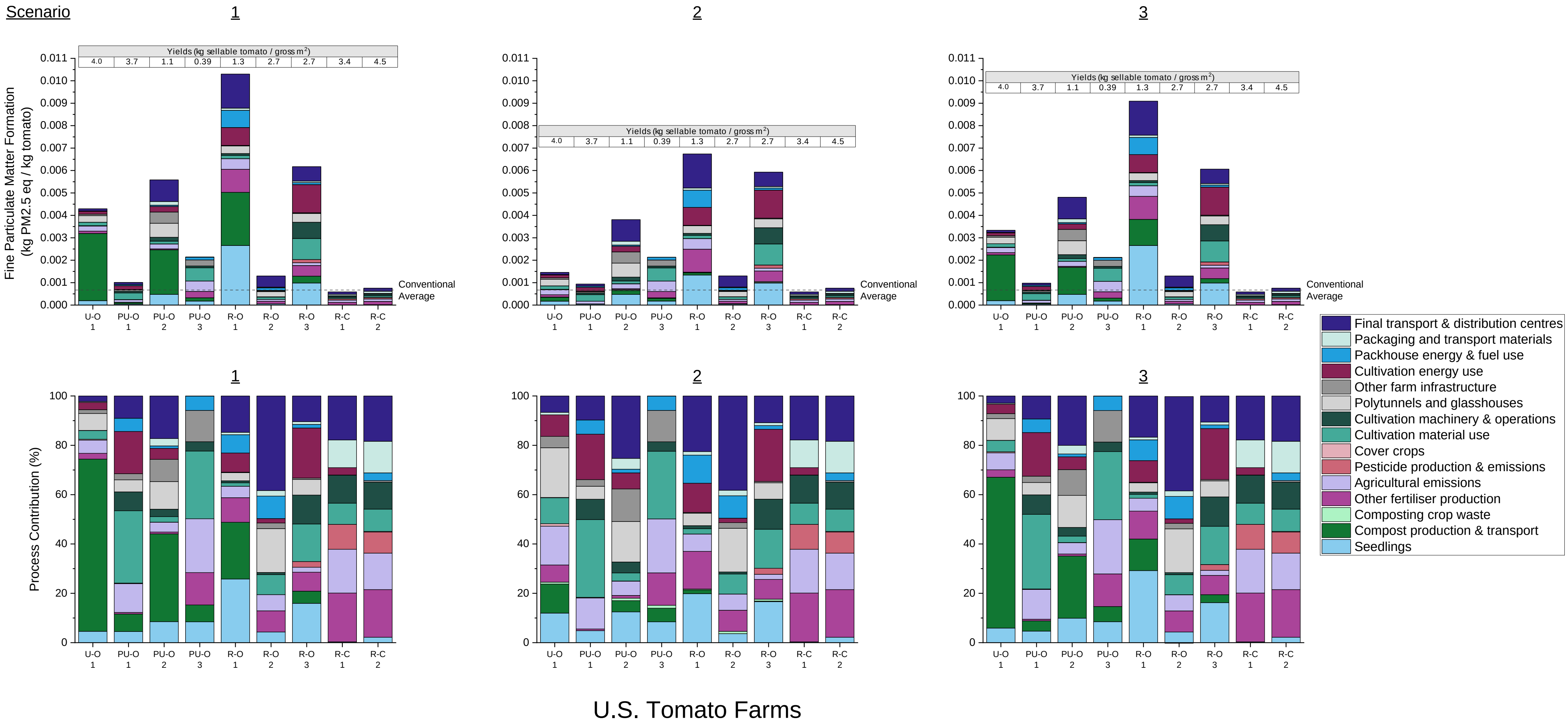

Figure 39 *– Fine particulate matter formation impacts and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg PM2.5 eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross* m^{-2}*).*

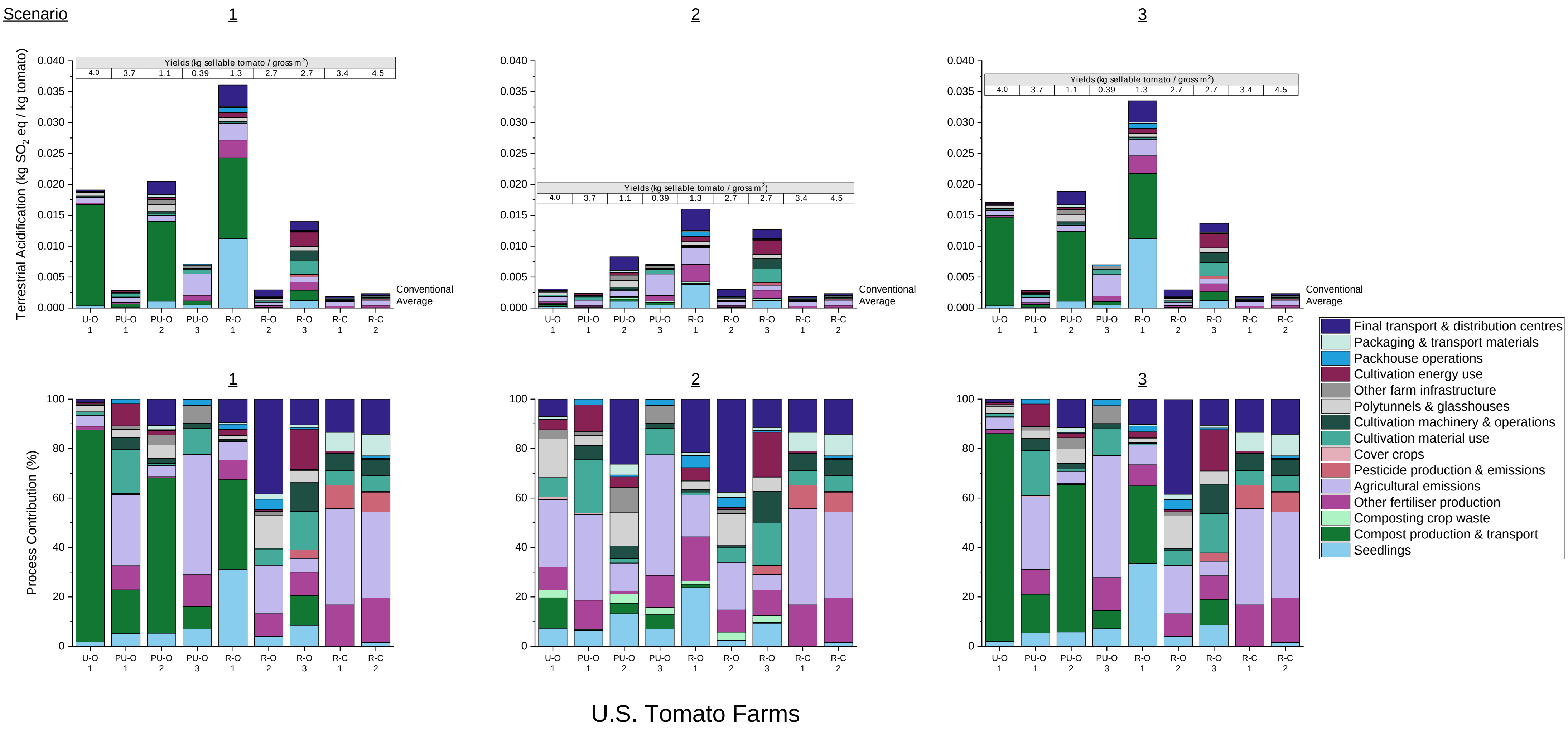

Figure 40 *– Terrestrial acidification impacts and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg SO_2 eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

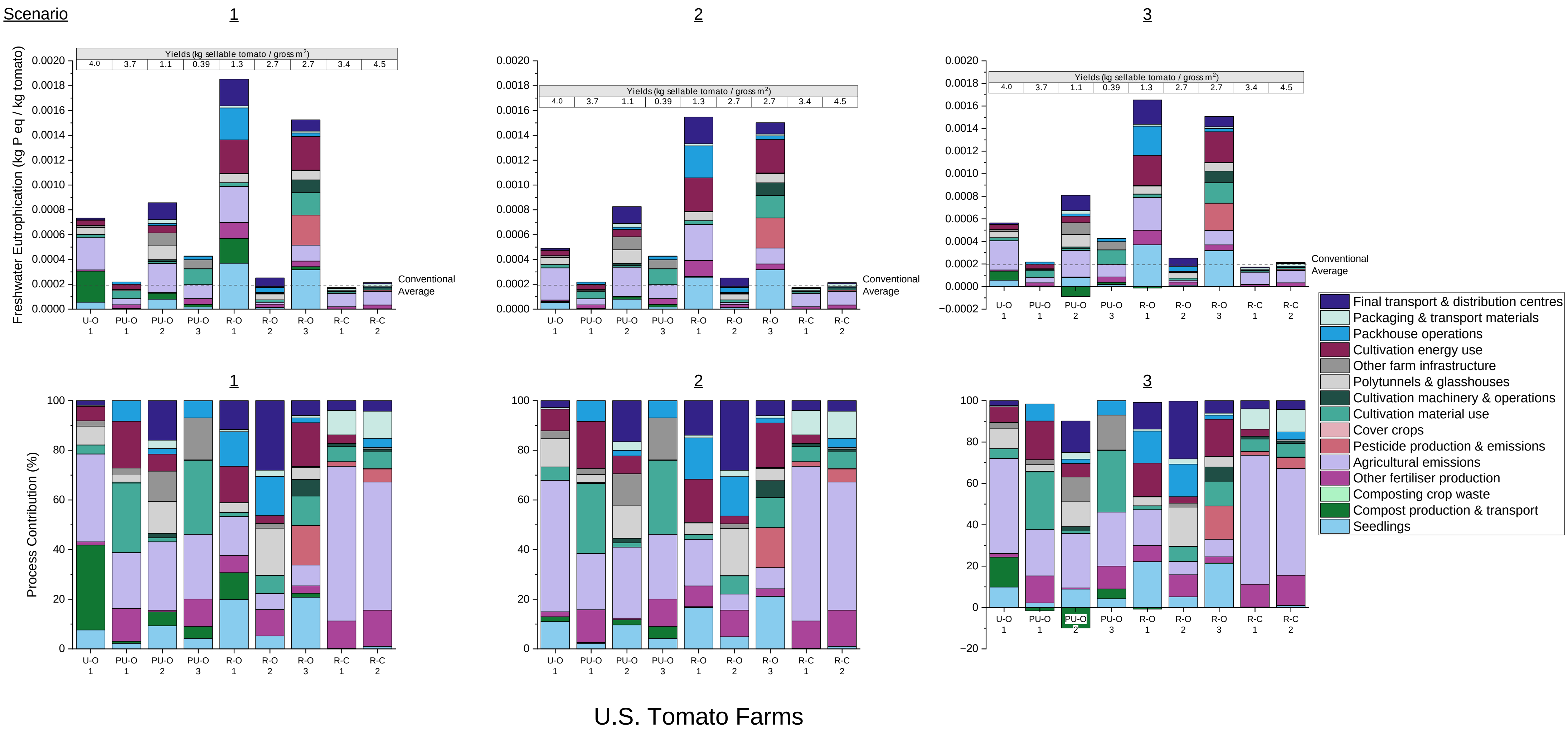

Figure 41 *– Freshwater eutrophication impacts and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg P eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

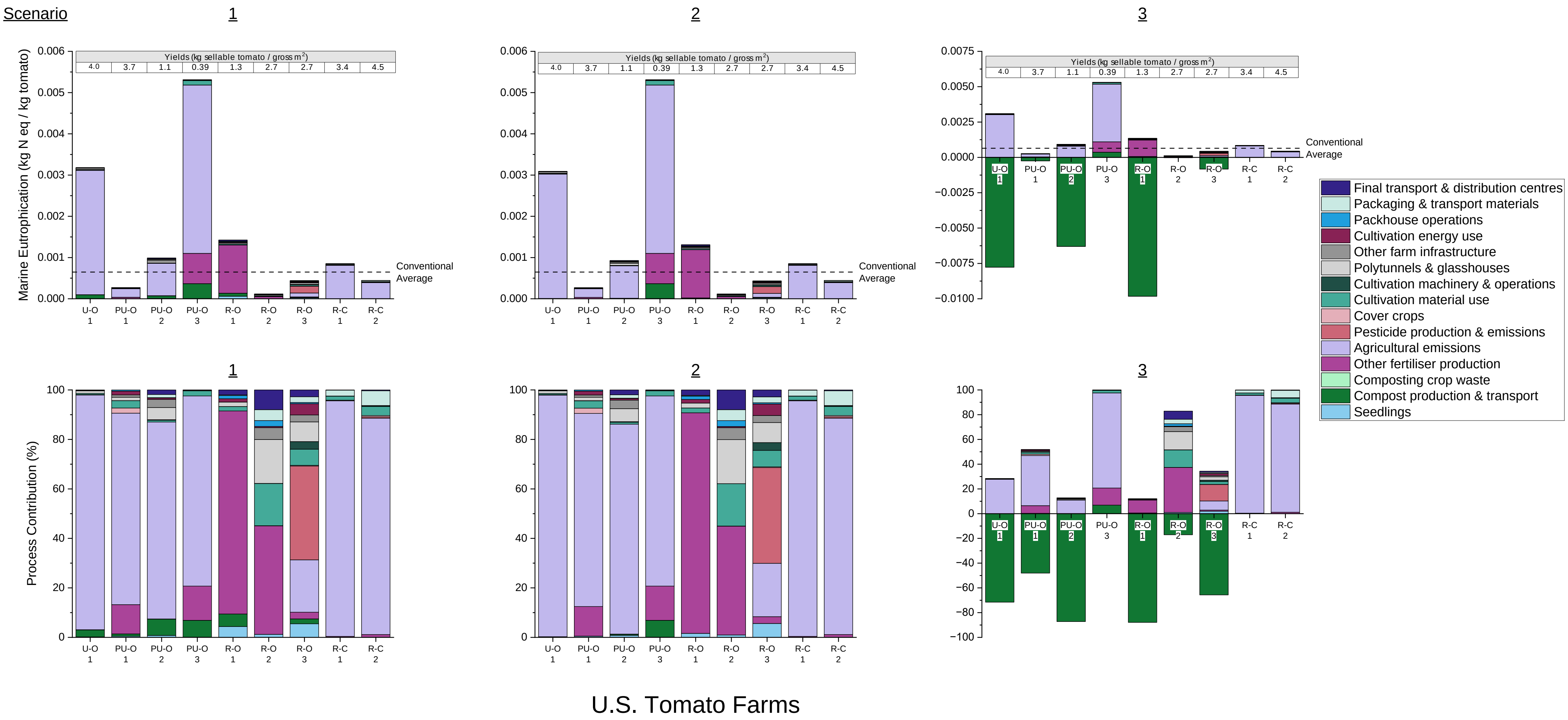

Figure 42 *– Marine eutrophication impacts and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in kg N eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

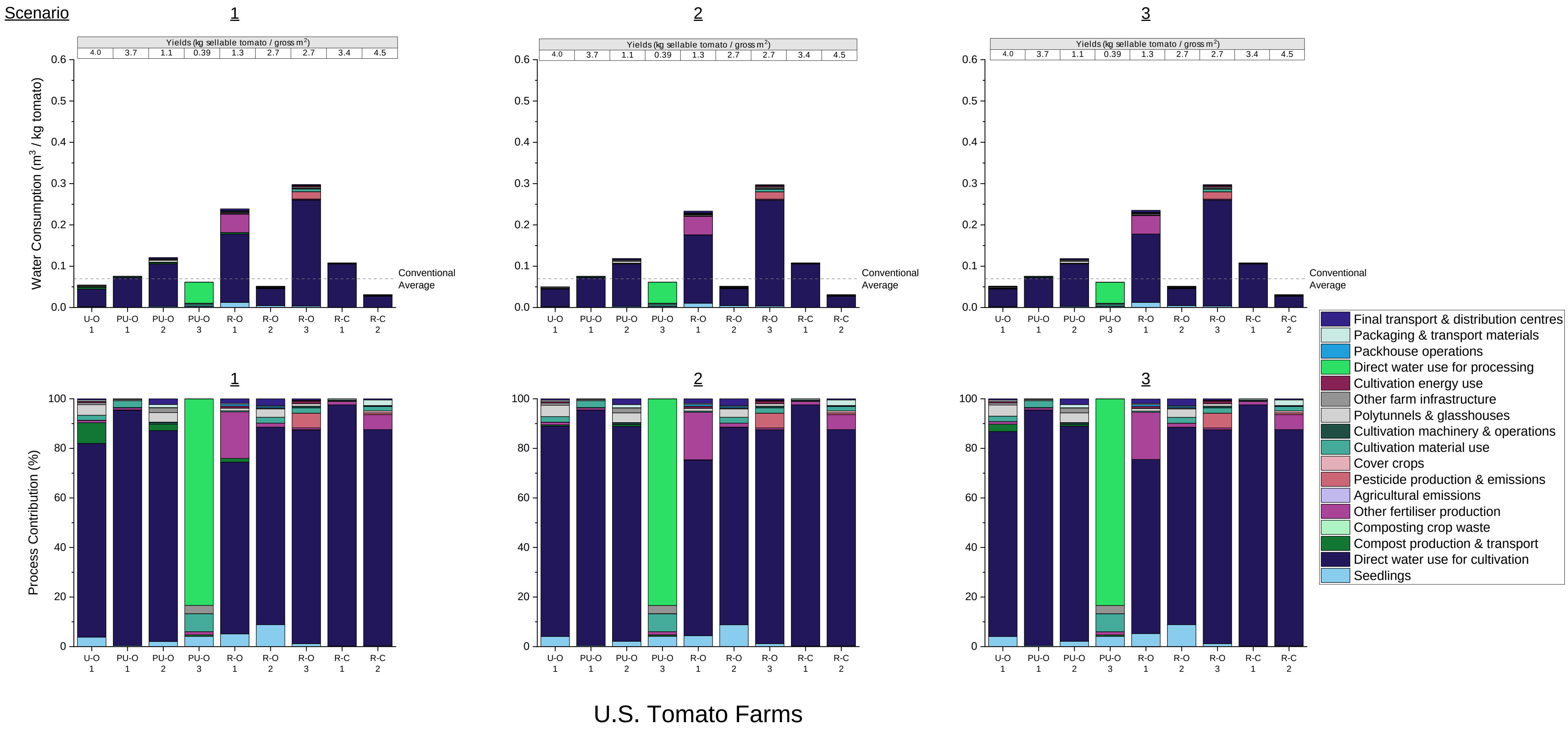

Figure 43 *– Water consumption impacts and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in m³ per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m⁻²).*

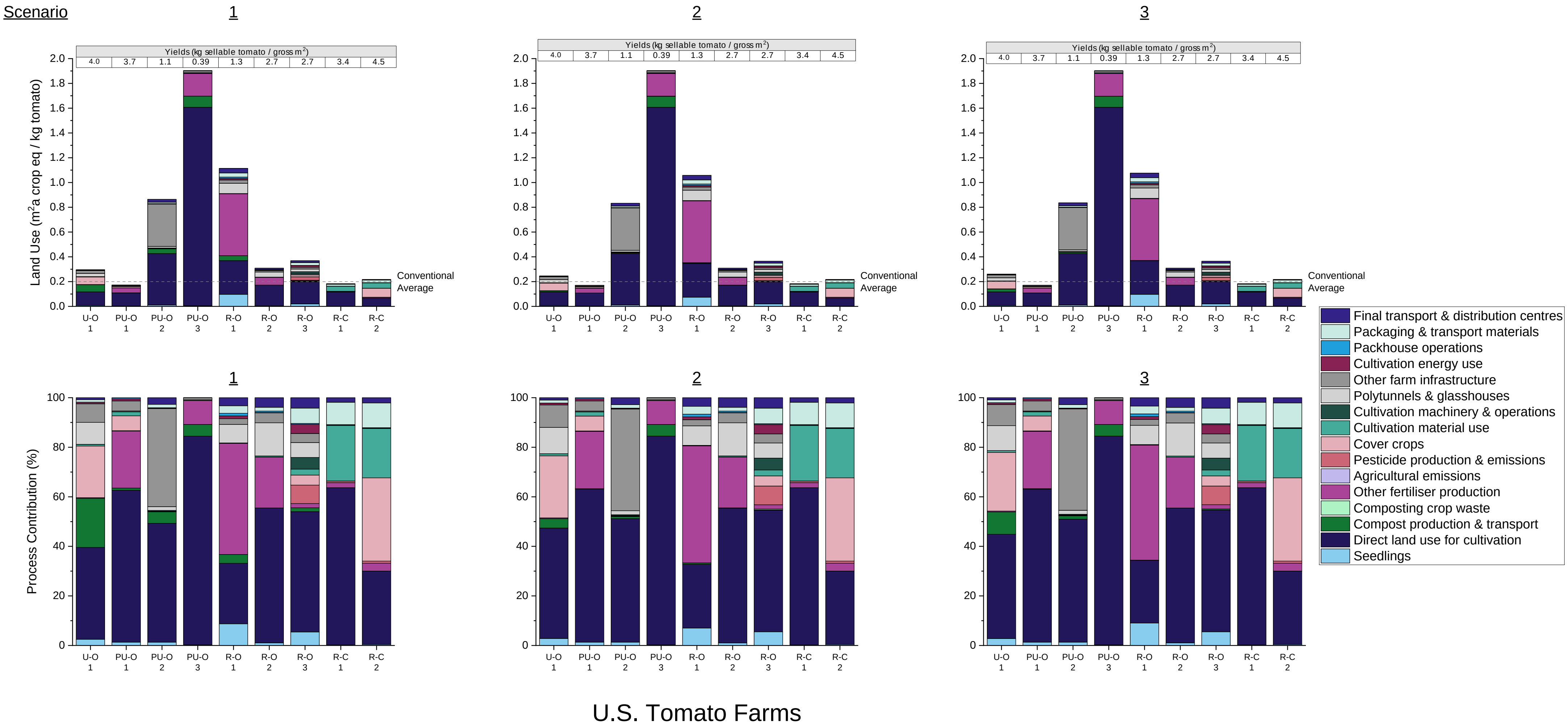

***Figure 44** – Land use impacts and contributing processes for individual U.S. tomato farms (n=9), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C) farms, with results from allocation Scenarios 1-3 displayed from left to right. The top row displays actual impacts in m^{2}a crop eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

3.1.2.4 UK Tomatoes

Figure 45 through Figure 51 display respective global warming, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, water consumption, and land use impacts across individual farms for UK tomato production. Trends between individual farms, and the main processes contributing to these trends, are discussed within this section.

3.1.2.4.1 Global warming, fine particulate matter formation, and terrestrial acidification

Global warming (Figure 45), fine particulate matter formation (Figure 46), and terrestrial acidification impacts (Figure 47) for UK tomato production in Scenario 1 are driven largely by a few main processes, unlike the wide variety of factors contributing to impacts for U.S. tomato production (Figure 38-Figure 40). For organic farms, this is compost production, as well as agricultural emissions, particularly for the latter two impact categories. For the conventional farm cases, contributions and relative impact levels vary more across the three impact categories but are mainly driven by cultivation energy use (from heating glasshouses), glasshouse infrastructure, and other farm infrastructure.

Throughout this data, more similarities in impact levels are observed between farms in the same farm category than seen for UK kale production. This is perhaps because growing practices for tomatoes are more similar among all UK organic farms, with tomatoes always grown in polytunnels using irrigation and with all but one farm using compost as an input. This is in contrast to UK kale production, where only some farms use compost as an input, and U.S. tomato production, where cultivation occurs both inside and outside, thus requiring different materials and infrastructure.

As nearly all organic farms use compost as an input for tomato production, differences in impact levels in Scenario 1 are largely driven by amounts of compost use. R-O-2 is the only organic farm that did not apply any compost, using solely manure for fertilisation, and this farm is thus seen to have either the lowest or second lowest global warming, fine particulate matter formation, and terrestrial acidification impacts in Scenario 1, rivalled by PU-O-4, which also used low amounts of compost. Impacts from compost use come from both the burdens of compost production and the agricultural emissions associated with its application. Global warming potential is mainly driven by compost production impacts, comprising anywhere from 36-80% of impacts for the organic farms using compost. This is why farms applying high levels of compost see large global warming impact reductions in Scenario 2. On the other hand, impacts for fine particulate matter formation and terrestrial acidification are driven by compost production as well as the agricultural emissions associated with compost (and other fertiliser) application. Together, these comprise 16-88% of fine particulate matter formation impacts and 46-95% of terrestrial acidification impacts among organic farms, thus showing the massive variation observed between farms due to varying levels of compost inputs.

Across these impact categories, impact levels are seen to generally decrease from urban to rural organic production, as was identified in the averaged results (Figure 17-Figure 19). This is because the more urban farms tend to use higher amounts of compost. Thus, the highest impacts are seen for U-O-1, U-O-2, PU-O-1, and PU-O-3, the four farms that apply the highest levels of compost per area. Higher compost use for these farms does not always result in higher yields, seen in particular for U-O-1 and PU-O-3, which have the lowest yields out of all farms. On the other hand, the rural organic farms (R-O-1 and R-O-2) and PU-O-4 tend to have the lowest impacts overall, as they use less (or no) compost by offsetting this with the use of manure. The rural organic farms also have some of the highest yields out of the organic farms. PU-O-2 also sees low impact levels similar to the rural organic farms, as this farm also used relatively low amounts of compost inputs per area, but also had a relatively low yield (1.7 kg m^{-2}) compared to most other farms.

Other contributors to impacts on organic farms are minor. Polytunnel infrastructure contributes more to global warming and fine particulate matter impacts, but this is still generally <15% for most farms. The use of cultivation materials is a much lower contributor than seen for U.S. tomato production, largely because all tomato production occurs in polytunnels rather than outdoors and thus does not require staking materials or steel cages for trellising, which contributed to the higher impacts from this category for the U.S. farms. Final transport is also less significant impact contributor for UK tomato farms than seen in the U.S., due to the smaller distances that food is transported, even for rural organic farms. For the urban farms, transport impacts are almost negligible.

The two cases of UK conventional production (using conventional energy sources or CHP) are seen to have more variable contributions of impacts across categories. Global warming potential impacts are driven mainly by the energy use from heating the glasshouses, accounting for 70% of total impact for both farm cases. The use of CHP slightly lowers global warming potential, but only by 10%. The high energy use for heating thus drives relatively high impacts for both cases of UK conventional production, which have the highest global warming potentials in comparison to all other organic farms. However, for fine particulate matter formation and terrestrial acidification, the contribution to impact from energy use is much lower. Thus, the conventional farm cases display relatively moderate impact levels, within the range of other organic farms. R-C-NG sees slightly higher contributions from cultivation energy use for these impacts, comprising approximately 25%, in contrast to R-C-CHP, where this accounts for <10% of impacts.

The major contributions to fine particulate matter formation and terrestrial acidification impacts for the conventional farm cases come from glasshouses and other farm infrastructure, which collectively constitute approximately 40-55% of total impacts. This is mainly from the infrastructure needed for the cold storage of crops, which is generally not seen for the organic farms, as most do not use a refrigerated cold store for tomatoes. Unlike for most other farms in this study, fertiliser production and agricultural emissions play less of a role in global warming, fine particulate matter formation, and terrestrial acidification impacts for UK conventional production, collectively comprising <11% of terrestrial acidification and fine particulate matter formation impacts and actually resulting in slightly negative contributions for global warming potential, due to negative contributions from CO_2 incorporated into crop biomass. Although the UK conventional farm has relatively high levels of N inputs per area compared to the other organic farms, levels are much lower per crop output due to the extremely high yields achieved through hydroponic production (33 kg m^{-2} vs. 3.0 kg m^{-2} average on the organic farms). This high yield thus drives down impacts seen from fertiliser production and application. Final transport is also not a significant contributor to impacts for UK conventional production (<4% for all three categories), despite the fact that crops from this farm are transported throughout the UK.

For global warming potential (Figure 45), all organic farms have lower impacts than the conventional farm cases, but more variation is seen for the other impact categories. For fine particulate matter formation (Figure 46), three organic farms have higher impacts, which are the farms applying the highest levels of compost per unit area (U-O-1, U-O-2, and PU-O-3). These farms also have higher impacts than the conventional farm cases for terrestrial acidification (Figure 47), in addition to PU-O-1.

Because of the contribution of compost production burdens to all three impact categories, large differences are observed between scenarios. In Scenario 2, the largest differences are seen for global warming potential, as compost production is the highest contributor to this impact category in Scenario 1. Once compost production burdens are removed, almost all organic farms are seen to have similar impact levels except for PU-O-3, for which impacts were still driven higher by seedling, material (from rainwater harvesting equipment), and transport burdens.

For fine particulate matter formation and terrestrial acidification, reductions in impacts are observed for all farms using compost in Scenario 2, but impact levels are less similar between organic farms than seen for global warming potential. This is because the agricultural emissions from compost and fertiliser application contribute more to these impact categories (especially terrestrial acidification) than for global warming potential; thus, variation between organic farms in Scenario 2 is driven mainly by differences in agricultural emissions, which is again tied to amounts of applied compost. The lowest impacts are therefore still seen on the farms using the lowest amounts of compost (PU-O-4, R-O-1, and R-O-2). For fine particulate matter formation, reductions from compost production drive impact levels for both urban farms lower than the conventional average, but otherwise basic trends remain unchanged for fine particulate matter formation and terrestrial acidification in Scenario 2.

In Scenario 3, the avoided burdens from compost production result in negative contributions and changes in trends for global warming potential. In particular, the farms applying the highest amounts of compost are seen to have the lowest global warming potentials, which are mainly the urban and peri-urban farms. Indeed, all organic farms except R-O-2, the only farm that did not apply compost, have overall negative global warming potentials. On the other hand, avoided burdens from compost do not significantly affect the fine particulate matter formation or terrestrial acidification categories, and thus trends between organic farms remain unchanged from Scenario 1 for these impacts.

For UK tomato production, there is also the consideration of the avoided burdens from the production of surplus electricity in CHP, which affects R-C-CHP. These avoided burdens result in only a slight decrease of global warming potential, which does not affect overall trends for this impact category. However, large negative contributions from avoided burdens are seen for the fine particulate matter formation and terrestrial acidification categories, driven mainly by avoided emissions associated electricity production via coal, even though this is a relatively low proportion of the UK energy mix (<8% for the Great British electricity grid in ecoinvent). These avoided burdens result in R-C-CHP having the lowest impact levels for these categories out of all other farms in Scenario 3, even resulting in an overall negative terrestrial acidification impact.

3.1.2.4.2 Freshwater eutrophication

Freshwater eutrophication impacts for UK tomato production are driven primarily by agricultural emissions (associated with P leaching) for organic farms, as seen in Figure 48. Unlike the previous impact categories (Figure 45-Figure 47), compost production is not a significant contributor to impacts. For conventional production, impacts are driven primarily by other farm infrastructure, as well as cultivation materials and agricultural emissions.

The differences that arise between organic farms are again mainly related to levels of compost and other fertiliser use, which drive impact contributions from agricultural emissions through P leaching. As for the previous three impact categories, the lowest impacts are observed for R-O-2, PU-O-4, and R-O-1 (in ascending order of impacts), the farms that apply some of the lowest levels of compost and other fertiliser inputs. Only PU-O-3 has impact levels higher than the conventional average, as a result of high P inputs from high levels of compost application. Unlike for U.S. tomato production, cultivation materials and energy use are not significant contributors to impacts for organic farms, due to both the lower levels of material and energy use and also because of the lower proportion of coal power in the Great British electricity grid.

Impact levels for the conventional production cases are largely similar and are driven primarily by other farm infrastructure rather than agricultural emissions. Indeed, farm infrastructure constitutes approximately 40% of total freshwater eutrophication impacts. This is mainly associated with the production of metal components for cool rooms. Agricultural emissions do contribute to impact, but to smaller extent (approximately 15% of impacts), with a similar contribution also seen from cultivation materials. This contribution is actually driven by the use of jute twine as opposed to plastic twine for trellising, which contributes to freshwater eutrophication from the fertiliser application associated with cultivating the jute plant.

As freshwater eutrophication impacts are not driven to a significant extent by compost production, no major differences in trends are observed in Scenario 2. On the other hand, the avoided burdens from the municipal solid waste stream do contribute to negative contributions in Scenario 3, as also seen for UK kale production (but less so for U.S. production). This is due to the avoided burdens associated with incineration within British municipal solid waste management. These avoided burdens thus result in negative contributions for all organic farms using compost, driving impact levels for all organic farms below that of R-C-NG. A negative contribution from the avoided burdens applied to surplus electricity is also seen for R-C-CHP in Scenario 3, again driven mainly by avoided electricity production via coal power. These avoided burdens result in this conventional farm case now having impact levels approximately one-third that of R-C-NG, although impacts for most organic farms (except PU-O-3) are still lower.

3.1.2.4.3 Marine eutrophication

As for other crop lifecycles, marine eutrophication impacts for UK tomato production are generally driven by agricultural emissions as a result of nitrate leaching from compost and fertiliser use (Figure 49). However, for nitrate leaching is assumed to be zero for many farms (U-O-1, PU-O-1, PU-O-2, PU-O-4) because these farms cultivate tomatoes indoors using drip irrigation, which is assumed to result in zero leaching as per IPCC guidelines (IPCC, 2019). The other organic farms also grow tomatoes in polytunnels, but use other types of irrigation (e.g., sprinklers), which are assumed to contribute to leaching. For the hydroponic production employed by the conventional farm, nitrate leaching is assumed to come only from the leachate generated by the open hydroponic systems, which comprise 50% of their total systems (see: Section 2.1.9.5.3 for more detail).

Because of these differences in assumptions of nitrate leaching based on irrigation practices, large differences in impact levels are observed between organic farms. Nitrate leaching is assumed to be non-zero only for U-O-2, PU-O-3, and R-O-2 (out of the organic farms). The prior two are seen to have the highest overall impact levels, at least 18x higher than the other organic farms, because these are also the farms that apply some of the highest levels of compost. Impacts for these farms come almost exclusively from agricultural emissions, accounting for >90% of impacts. Although R-O-2 also has contributions from agricultural emissions through nitrate leaching, in real terms this contribution is much smaller due to the much lower amounts of applied N (through manure only), as well as the lower amounts of irrigation applied through non-drip systems, as this farm used both drip and sprinkler irrigation at different stages of crop growth.

All organic farms besides U-O-2 and PU-O-3 have marine eutrophication impact levels lower than the conventional average by at least 5x. As nitrate leaching is assumed to be zero on all these farms (except R-O-2), contributions to impact come mainly from compost production, polytunnel infrastructure (seen most for farms using wood-framed polytunnels), other infrastructure, and, to a smaller extent, cultivation materials. R-O-1 has the highest contribution from cover crops, due to the estimated N leaching associated with cover crop seed production.

Impact levels between the conventional farm cases are similar and are higher than most organic farms that did not have nitrate leaching impacts, but approximately 3x lower than the highest impacting organic farms (U-O-2 and PU-O-3). Impacts for conventional production are driven primarily by agricultural emissions through nitrate leaching, accounting for approximately 50% of total impact. Other contributions come from farm infrastructure and the use of cultivation materials, each contributing to approximately 18% of impacts. This is again driven by infrastructure in the cool room and the use of jute twine (and thus the associated fertiliser use for jute plant production).

Comparing across scenarios, little change is observed between Scenarios 1 and 2 since compost production is not a major contributor of impacts for marine eutrophication, as for freshwater eutrophication. However, negative contributions are again seen in Scenario 3 from the avoided burdens applied to both compost production and CHP. The avoided burdens applied to the composting process arise mainly from the avoided ammonium and nitrogen emissions associated with landfilling. This results in all farms that apply compost (which includes all organic farms except R-O-2) having overall negative marine eutrophication impacts in Scenario 3. The farms applying the highest levels of compost (U-O-1, U-O-2, PU-O-1, and PU-O-3) thus have the lowest overall impacts. For R-C-CHP, the avoided burdens applied to surplus electricity from CHP reduce impacts but to a much lesser extent than seen for freshwater eutrophication. Thus, impact levels between R-C-NG and R-C-CHP are still similar in Scenario 3.

3.1.2.4.4 Water consumption

Water consumption for UK tomato production is driven mainly by direct water use for irrigation in the cultivation phase, with no direct water use seen during processing for any of the farms (Figure 50). Water use during cultivation accounts for >75% of impacts across all organic farms. The exception is PU-O-3, which does not have any impacts applied from water consumption. This is because PU-O-3 uses exclusively water captured via rainwater harvesting, which does not contribute to water consumption impacts in ReCiPe 2016. This is also consistent with crops cultivated outdoors, for which precipitation is not estimated within the water consumption category. Thus, PU-O-3 is seen to have the lowest water consumption impact out of all farms. For the conventional farm cases that produce tomatoes hydroponically, water use during cultivation accounts for approximately 50% of impacts, with other contributions also seen from the use of cultivation materials (25%) and other farm infrastructure (10%). For the prior, this again comes mainly from the water use associated with cultivating the plants used to make jute twine.

Although it is assumed that hydroponic cultivation would result in much lower water consumption levels than other soil-based cultivation, as this is generally seen as a main benefit of hydroponic production, this is not actually the case for UK tomato production. This is in part because of the added water consumption associated with the use of jute twine on the conventional farm, but even without this contribution, several organic farms have lower or similar direct water uses during cultivation as seen for hydroponic production (e.g., R-O-1, R-O-2, PU-O-2, and PU-O-4). Higher water use is generally observed on the urban farms, which both have water consumptions above the conventional average. Additionally, PU-O-1 emerges as an outlier, with water consumption levels approximately 4x higher than the urban farms, simply due to the use of higher irrigation rates.

As compost production is not a major contributor to water consumption impacts, little change is observed across scenarios, especially for Scenario 2. In Scenario 3, the avoided burdens applied to compost production do result in negative contributions, but this does not largely change results between organic farms. The exception is for PU-O-3, which has such a low water consumption in Scenario 1 (due to not having any contribution from direct water use in cultivation) that the avoided burdens applied to compost production result in this farm having an overall negative water consumption impact in Scenario 3. For R-C-CHP, the avoided burdens applied to surplus electricity also result in negative contributions but these are minor, with impact levels reduced by only 25% compared to R-C-NG.

3.1.2.4.5 Land use

Land use impacts are displayed in Figure 51. As for other crop lifecycles, impact differences between farms are largely related to yields. This is why the lowest impacts are seen for conventional production, as conventional yields are drastically higher than organic yields due to year-round hydroponic production (33 kg m^{-2} compared to the organic average of 3 kg m^{-2}). The highest impacts are seen for U-O-1, the farm with the lowest yields; thus, this driven mostly by higher cultivation land use, but an additional contribution comes from polytunnel and other farm infrastructure (39%), mainly due to the use of wood in these structures. Contributions from infrastructure are also seen for U-O-2 and R-O-1. Indeed, although R-O-1 has the highest yield out of the organic farms (5.8 kg m^{-2}), it does not have the lowest overall land use because of contributions from infrastructure and cover crops.

Because of the very high yields achieved by conventional production, direct land use from cultivation actually accounts for a relatively small proportion of total impact (approximately 15%) for the conventional farm cases. Other land use contributions come from cultivation materials (35-40%), again associated with twine production, as well as farm infrastructure and packaging, which each account for approximately 12% of impacts. R-C-NG also sees an additional contribution (approximately 9%) associated with energy use during cultivation that is not seen for R-C-CHP, and thus the latter has a slightly lower overall land use.

When comparing across scenarios, negligible change is observed for the organic farms, as land use during compost production is not a major contributor to impact and the avoided burdens applied in Scenario 3 are minor. However, R-C-CHP sees a large negative contribution from the avoided burdens applied to surplus electricity production during CHP, resulting in a negative overall land use impact. This is mainly driven by the avoided burdens of biomass energy (and thus land use associated with biomass production) as part of the Great British national grid.

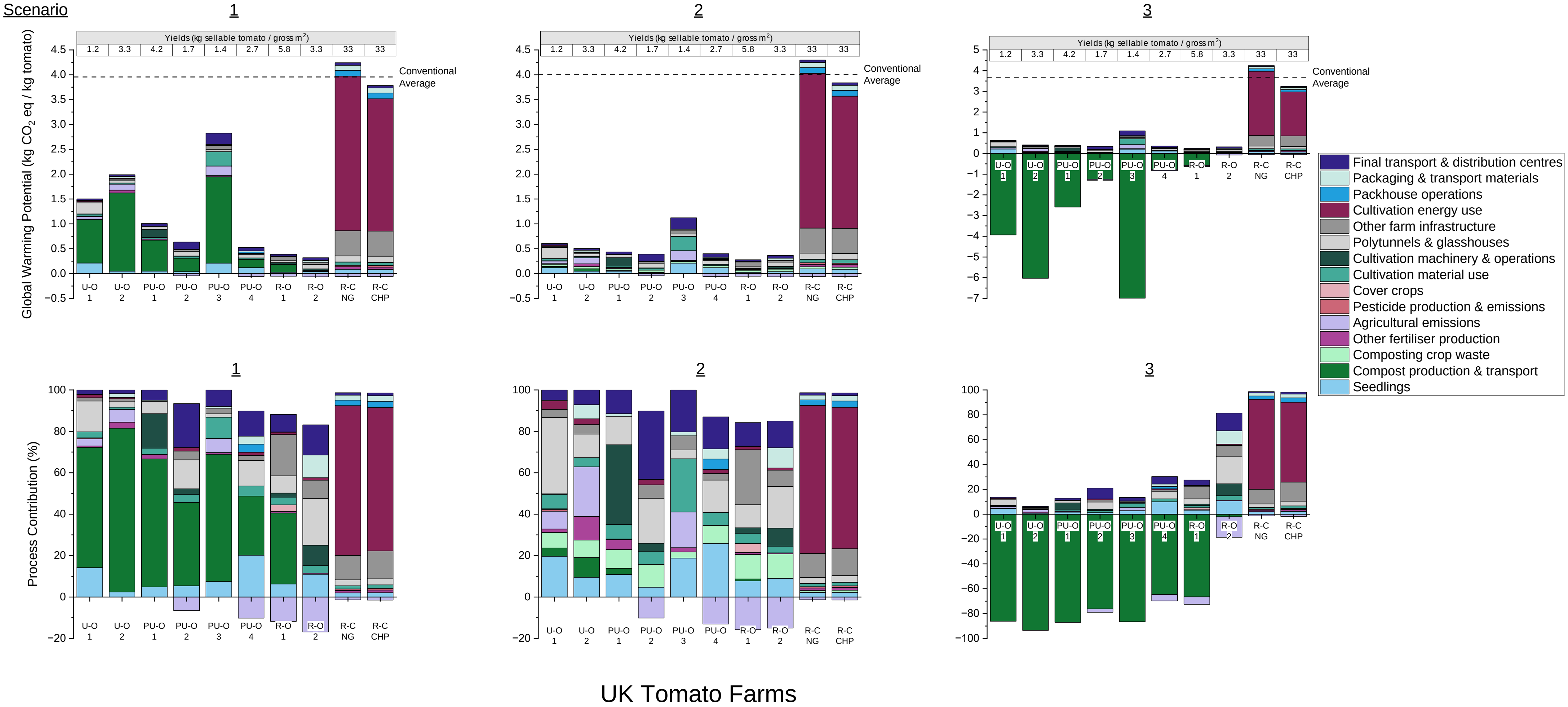

Figure 45 *– Global warming potential and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in kg CO_2 eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

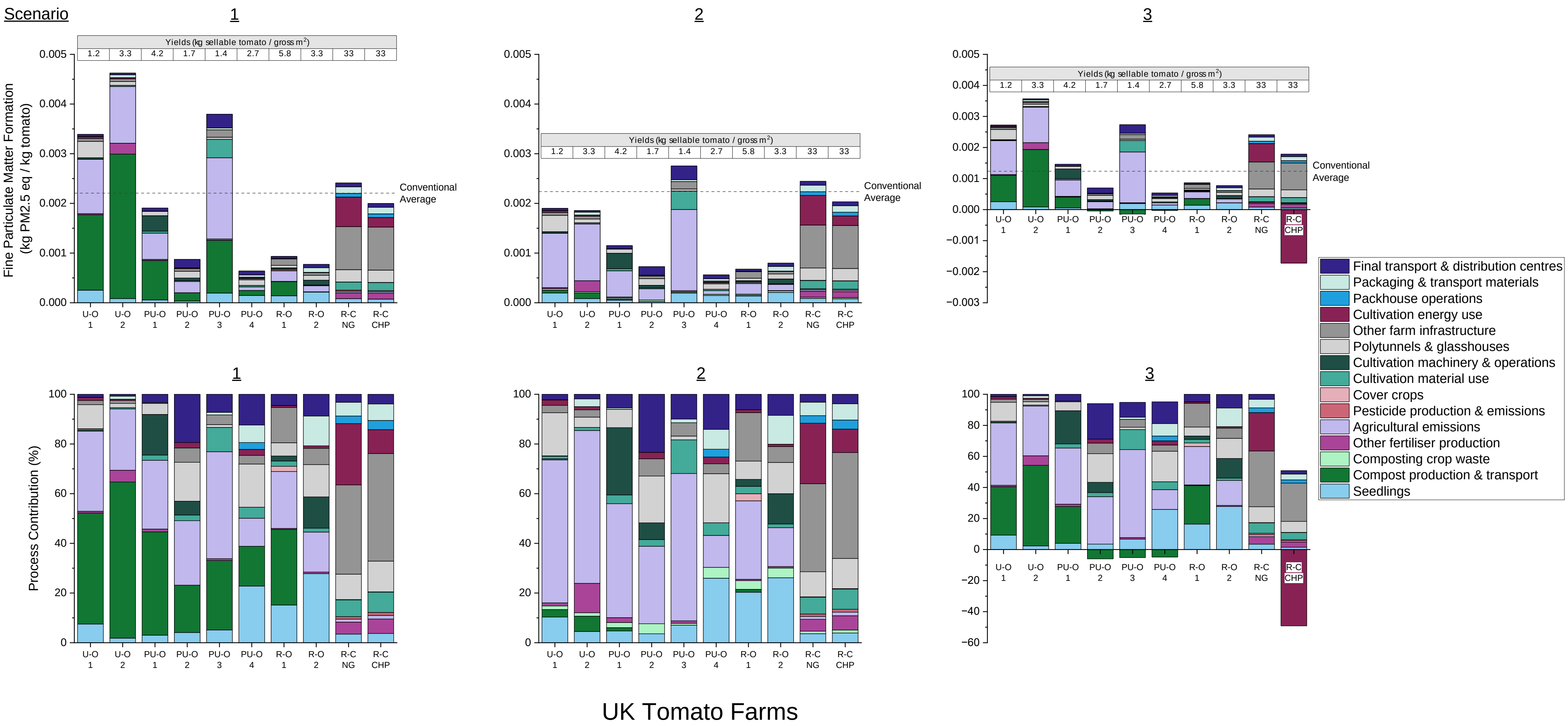

Figure 46 *– Fine particulate matter formation impacts and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in kg PM2.5 eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross* m^{-2}*).*

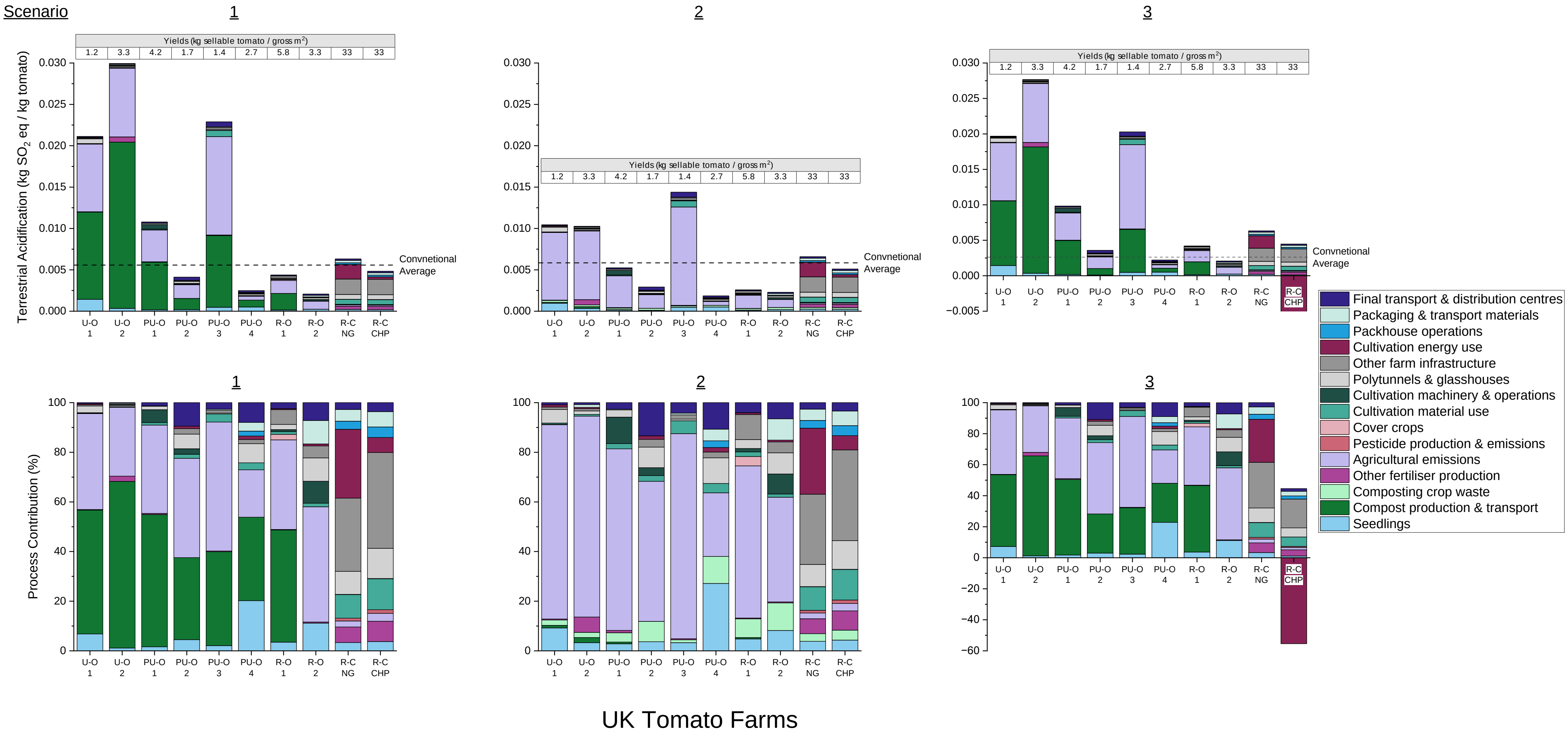

Figure 47 *– Terrestrial acidification impacts and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in kg SO_2 eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

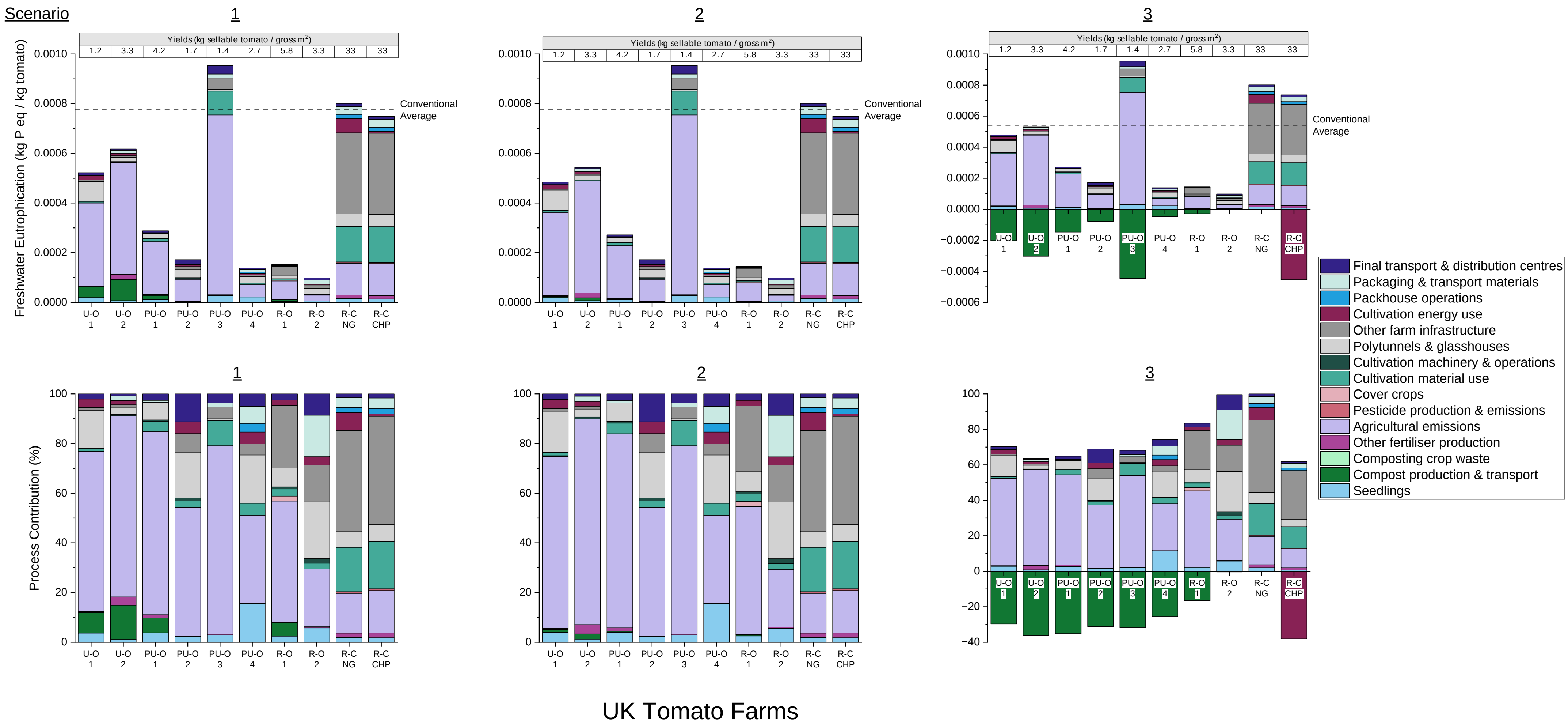

***Figure 48** – Freshwater eutrophication impacts and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in kg P eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

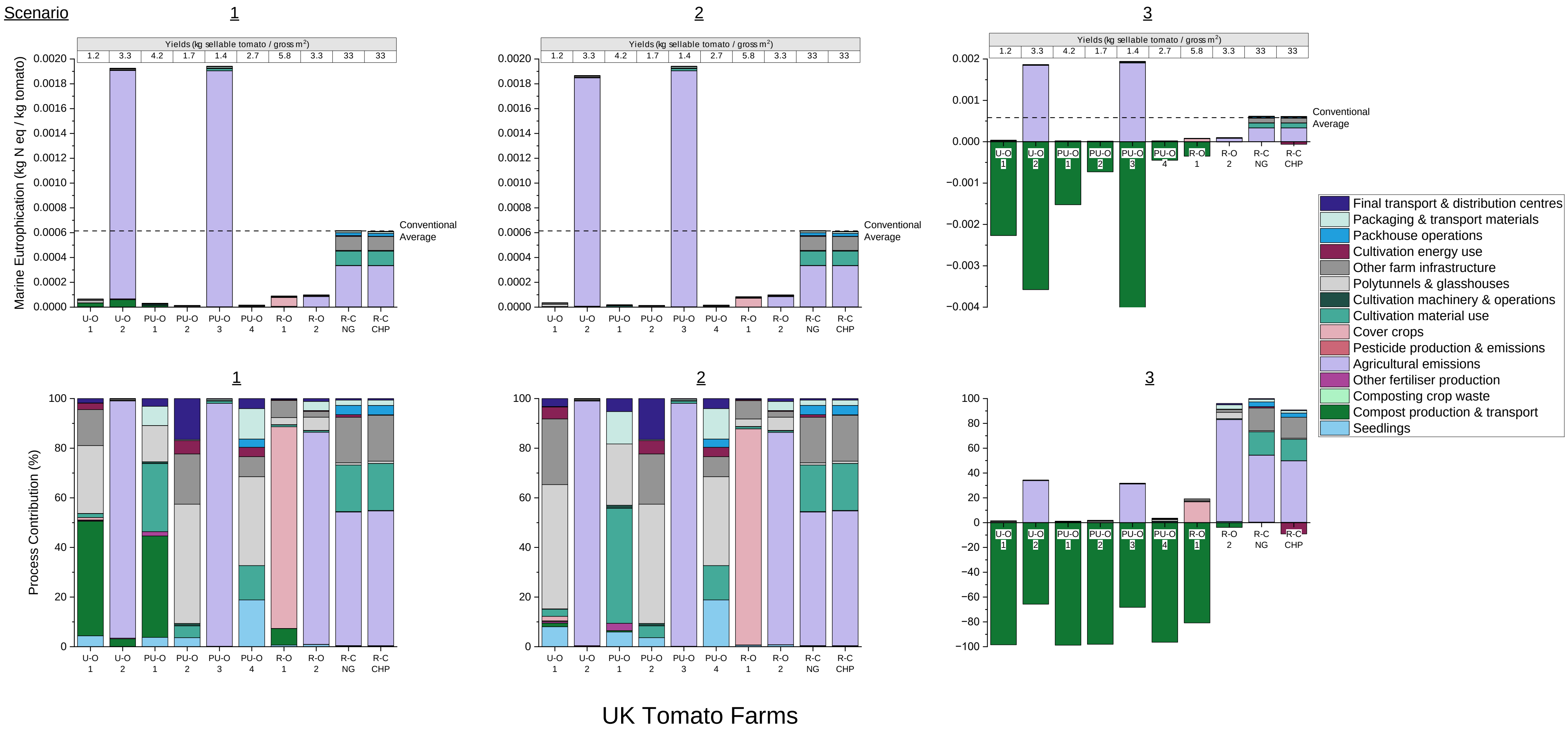

Figure 49 – Marine eutrophication impacts and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in kg P eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).

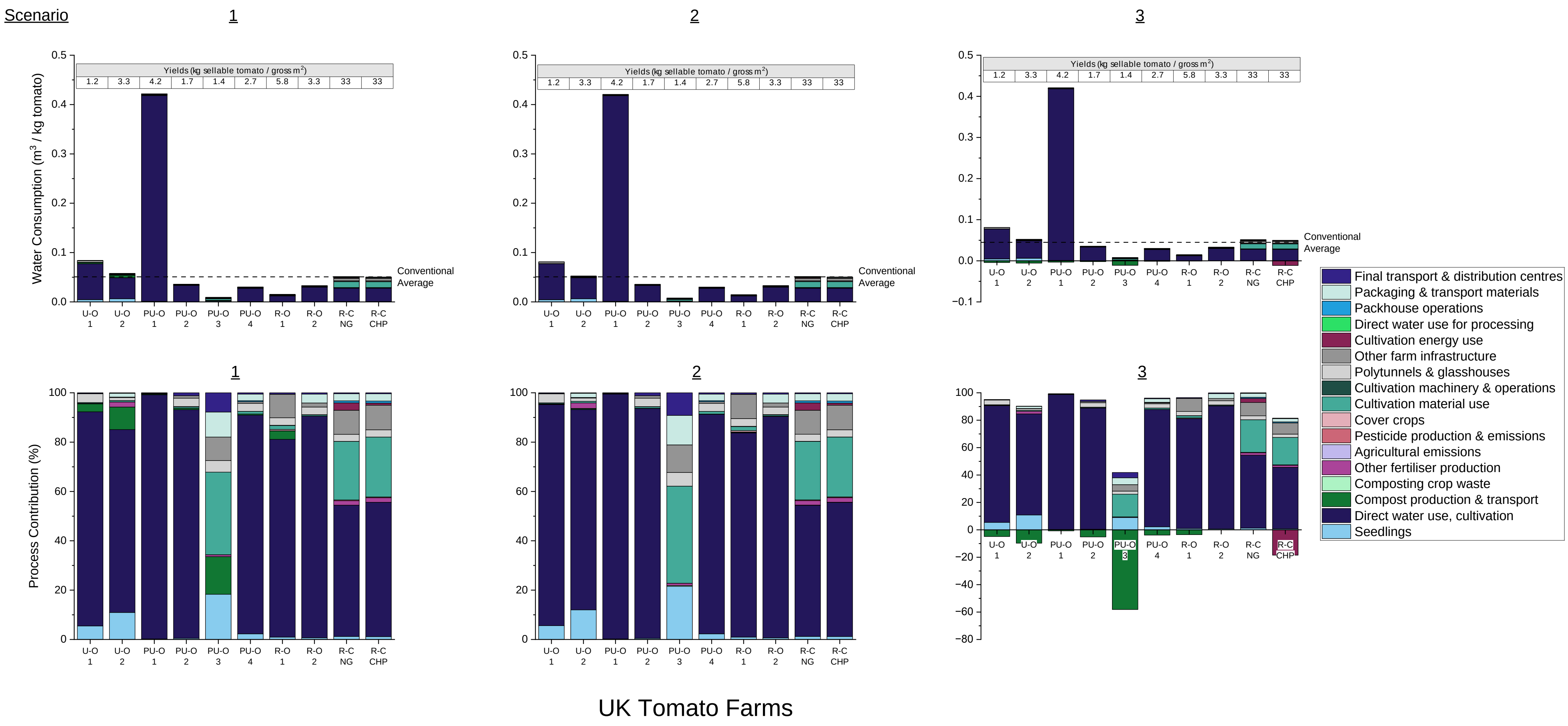

Figure 50 *– Water consumption impacts and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in m^3 per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

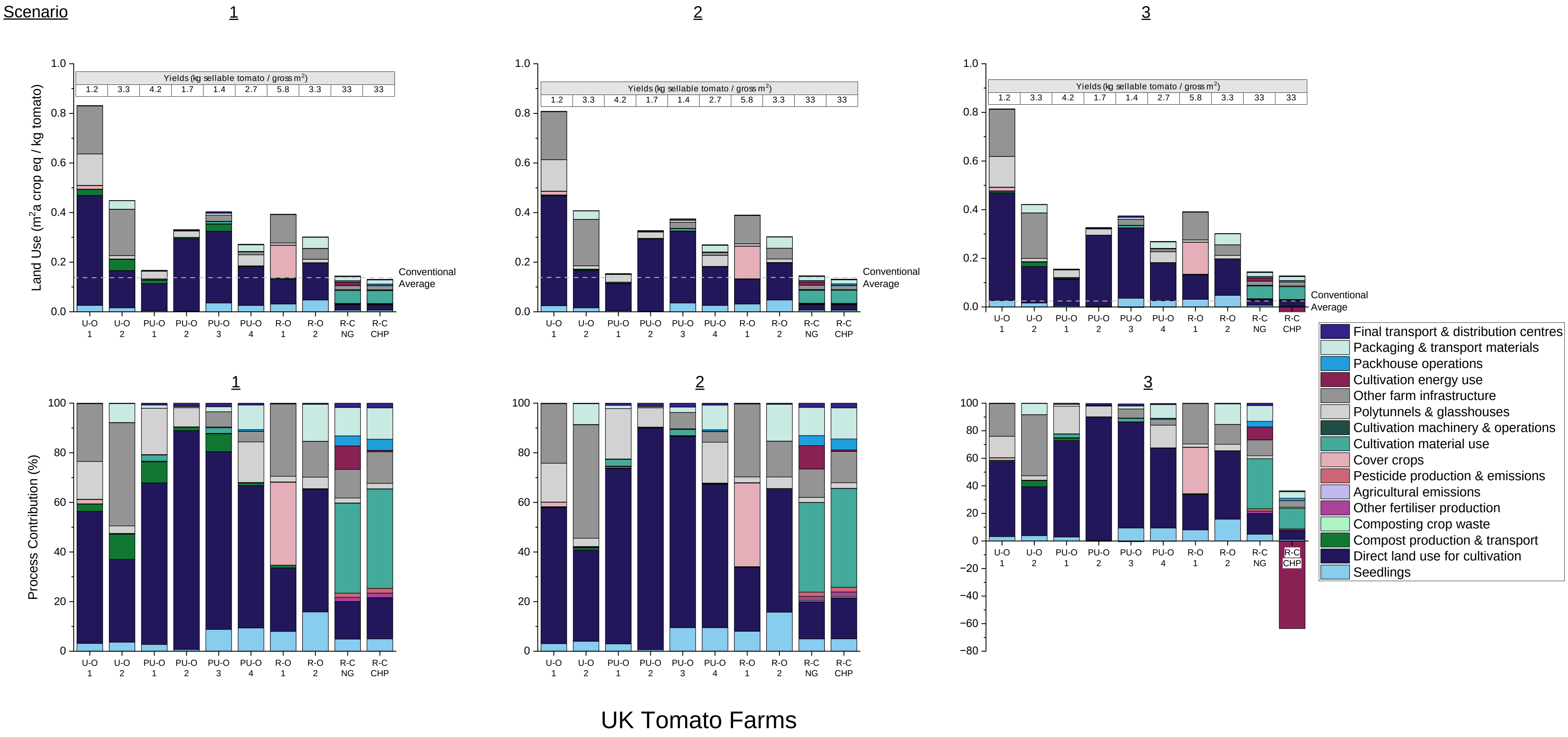

***Figure 51** – Land use impacts and contributing processes for individual UK tomato farms (n=10), including urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and two cases of rural conventional (R-C) production, based on energy type used (R-C-NG and R-C-CHP). Results from allocation Scenarios 1-3 are displayed from left to right. The top row displays actual impacts in m^{2}a crop eq per kg tomato transported to the final point of sale, with a dotted line designating the average impact level of the conventional farms. The bottom row displays process contributions to total impact out of 100%. Different colours represent the contributions from different processes to total impact, as designated in the legend. Tables within the top row provide yields for each corresponding farm, in kg sellable tomato (gross m^{-2}).*

3.1.2.5 Comparisons across countries and crop types

Considering the results presented for tomato and kale lifecycles on farms in the U.S. and UK (Figure 24-Figure 51), this section aims to highlight some of the overarching trends and differences observed between crops and countries.

Comparing organic production in the U.S. and UK, it is generally seen that a wider array of processes contributed to impacts on U.S. organic farms, especially for global warming potential, fine particulate matter formation, terrestrial acidification, and freshwater eutrophication. This is due to several reasons. For one, U.S. organic farms tended to use a wider amount of other fertilisers in addition to compost (e.g., seaweed fertilisers, feather meal, pelleted manure, and seed meals), whilst UK organic farms often used just compost or manure as the main fertiliser input. In addition, U.S. organic farms often saw higher contributions from energy use than in the UK. In one respect, this is because of the higher amount of coal power in the southeast U.S. energy mix, which contributes particularly to fine particulate matter formation (Figure 25 and Figure 39) and freshwater eutrophication impacts (Figure 27 and Figure 41). However, this is also because of the generally lower energy uses seen on UK farms, especially for kale production. This is because the majority of farms do not irrigate kale in the UK, or irrigate only intermittently, and also because many of the organic farms do not use cold storage for kale or only use it for a small portion of the year, due to the colder UK climate. For U.S. tomato production, several organic farms are also seen to have higher cultivation material uses for trellising than in the UK. This is because some organic farms grow tomatoes outdoors in the U.S., thus requiring steel cages or posts for trellising, whilst all tomatoes are grown inside polytunnels in the UK.

On the other hand, UK organic farms generally have higher contributions to impacts from cultivation machinery and operations, particularly for kale production. Although tractors are used across several organic farms in both countries, UK organic farms tend to use tractors for a higher number of operations. For example, many UK organic farms use tractors for a combination of ploughing, tilling, harrowing, hoeing, planting, and mowing and rolling cover crops, whilst in the U.S., tractors are used mainly for just one cultivation process (e.g., tilling or ploughing), bed making, and mowing cover crops. Additionally, U.S. farms tend to use only one tractor for all operations, whilst the organic farms using tractors in the UK tend to have several and may even use contractors.

In contrast, more similarities are observed between the processes contributing to impacts for conventional farms in the U.S. and UK. Fertiliser production and agricultural emissions are generally the main impact drivers for most categories, with the exception of UK conventional production. This case, as the only case of hydroponic, soilless production, sees much lower contributions from fertiliser production and agricultural emissions per kg crop produced, likely because of the extremely high yields per applied N seen for this farm. Other differences between countries are seen with contributions from infrastructure (e.g., sheds and packhouses), which are generally higher for the UK conventional farms than the U.S. farms. UK farms tend to have large-scale, steel warehouses on site to use for crop storage and packaging. In contrast, the U.S. farms tend to have more simple, wooden shed structures, and generally do not store crops or have very low storage times. In many cases, crops are packed directly into lorries for transport. This thus reduces infrastructural and storage burdens for

U.S. farms, meaning that contributions from packhouse operations (and energy use) tend to be larger for the UK than the U.S., the opposite of that seen for organic farms.

Across both rural organic and rural conventional farms, it is also clear that final transport contributes more to impacts (in real terms) in the U.S. than in the UK. Overall, this is because produce travels much larger distances in the U.S. due to the larger scale of the country, as opposed to the UK. For the conventional farms in Georgia, produce might travel across the entire East coast, with transport distances per kg crop in the range of 617-1,280 km (one way) compared to only 176-455 km for UK conventional farms, which also distribute across the country. The rural organic farms in the U.S. also travel longer distances to final points of sale than seen for the UK rural organic farms, because of the different urban make-up of Georgia versus England. Georgia has one main urban core surrounding the capital city (Atlanta), where the majority of people live; thus, the rural organic farms in Georgia all travel into the capital city (Atlanta) to sell through major farmers' markets and restaurants. On the other hand, England is composed of a wider array of smaller urban areas, and thus peri-urban and rural organic farms can travel into their nearest neighbouring town or city to sell produce (e.g., Bristol, Manchester, Sheffield, and Newcastle), rather than having to sell through one major capital city like London.

Comparing results between crop lifecycles, kale production on organic farms is generally seen to have higher contributions from cultivation machinery and operations in comparison to tomato production, particularly in the UK. In contrast, tomato cultivation generally has higher contributions from fertilisers and agricultural emissions as well as cultivation materials, due to the higher nutrient requirements of this fruiting crop and the need for trellising. Higher contributions are also seen from energy use in the packhouse for kale versus tomatoes on U.S. farms, since kale requires cold storage and generally tomatoes are often only cooled slightly using fans, if at all. Overall, this data suggests that crop type, local context, and specific management practices all influence and determine environmental impacts seen on individual farms.

3.1.3 Sensitivity analysis

In addition to the three allocation scenarios that have been presented in the previous results, additional analyses were also performed to understand the sensitivity of the results to certain assumptions and decisions made within the LCA. In particular, three main areas were investigated based on the uncertainties present in these areas within literature. These include: additional considerations for compost, nitrate leaching models, and soilless system emissions. Although the allocation scenarios (Scenarios 1-3) have previously investigated differences in compost allocation, further sensitivity analyses have been performed on compost due to its importance as a main impact contributor on organic farms. In particular, the exclusion of composting (fugitive) emissions and also the inclusion of carbon sequestration benefits have been tested separately to see how this influences results. For nitrate leaching and associated N_2O emissions, a wide range of models and emission factors exist to estimate leaching in agricultural soils; thus, four additional models have been tested in comparison to the default model (de Willigen 2000 model). Finally, several sensitivities have been performed on the soilless systems (for UK conventional tomato production), particularly regarding different air and leaching emissions, due to the vast uncertainty regarding emissions from soilless systems in LCA literature.

All sensitivity analyses have been performed in Scenario 1 only, as this is considered the default scenario; however, in some cases, the results from Scenarios 2 and 3 have been included for comparison purposes. When this is the case, it should be noted that these are the original results from these Scenarios (i.e., additional sensitivities have not been run within these scenarios). Results are presented with plots displaying new impact levels generated from the sensitivity analysis, along with side-by-side plots showing the percent change in impact level from the original value (in Scenario 1). Original values are reproduced in both plots to allow for comparisons. Finally, results have been presented for both countries and crop lifecycles using farm category averages as well as displaying new results for individual farms. It should be noted that most sensitivity analyses only affect certain impact categories, and thus only the affected impact categories are re-presented.

3.1.3.1 Composting emissions and carbon sequestration

In addition to the allocation scenarios, further analyses have been performed on the composting process as this is such a significant impact driver for organic farms. Two considerations have been tested. This includes 1) the complete exclusion of emissions associated with the composting process (i.e., fugitive emissions from the compost pile) and 2) accounting for carbon sequestration benefits from compost application. The prior has been due to the degree of uncertainty around whether significant emissions from the composting process actually occur; in many cases, emissions depend on the management of the compost piles, and thus some studies do not certain composting emissions or any at all by assuming compost piles are properly managed (Smith *et al.*, 2001; ICF Consulting, 2005; U.S. EPA, 2006; Recycled Organics Unit, 2007; White, 2012). (U.S. EPA, 2006; Sánchez *et al.*, 2015). In terms of carbon sequestration, several LCA studies and carbon assessment frameworks have included estimations of C storage from compost application (U.S. EPA, 2006; Recycled Organics Unit, 2007; Blengini, 2008; White, 2012; Saer *et al.*, 2013). Thus, this has been tested for here by using an average carbon sequestration value of 0.101 kg CO_2-eq (kg compost wet weight)$^{-1}$ as estimated through the consideration of several literature sources by (Boldrin *et al.*, 2009). This value, listed as 'C sequestration average' within the results, depicts the amounts of carbon expected to still be bound in the soil after 100 years. This is relevant to this LCA because global warming potential is considered on the 100-year timeframe within the methodology used (ReCiPe 2016, Hierarchist).

Some studies also attempt to account for additional carbon benefits from compost application besides direct C storage, such as increased soil water retention and reduced erosion. In particular, Saer *et al.* (2013) accounts for these indirect benefits of compost, as well as direct C storage, by using a value of 0.675 kg CO_2-eq (kg compost)$^{-1}$. This value has thus been tested as a 'carbon sequestration maximum' in the sensitivity analysis. However, it should be noted that there is high uncertainty around how these benefits should be quantified, and the time scale used within this carbon value. Because of these reasons, this carbon sequestration value would not be implemented in this LCA, but is still included within the sensitivity analysis to facilitate a critique of carbon sequestration benefits and comparisons with other studies using this value. Section 2.1.7.6.5 provides further detail about C sequestration in compost and values used in other studies.

The exclusion of composting emissions affects only the global warming, fine particulate matter formation, terrestrial acidification, and stratospheric ozone depletion impact categories, with the prior three presented within these results. The inclusion of carbon sequestration benefits, expressed as CO_2-eq within the soil, only affects global warming potential results. Thus, both cases are presented within global warming potential results for U.S. kale production (Figure 52), UK kale production (Figure 54), U.S. tomato production (Figure 56), and UK tomato production (Figure 58). For fine particulate matter formation and terrestrial acidification, only the exclusion of composting emissions are considered, as displayed for U.S. kale production (Figure 53), UK kale production (Figure 55), U.S. tomato production (Figure 57), and UK tomato production (Figure 59). Results from allocation Scenarios 2 and 3 are also included for comparison and reference purposes, but the sensitivity analyses (i.e., exclusion of composting emissions and inclusion of carbon sequestration benefits) have only been applied to Scenario 1. Original values from Scenario 1 are also included as the default value, and thus percent changes have been calculated from this value, where a negative value indicates a decrease, and a positive value indicates an increase. Finally, it should be noted that these sensitivity analyses only affect the farms that use compost as an input, which are only organic farms. Conventional farm values are presented in the results for reference but are unchanged.

The exclusion of composting emissions results in lower global warming potential, fine particulate matter formation, and terrestrial acidification impacts for all farms using compost as an input, with reductions in average impacts for organic farm categories ranging anywhere from 2-56%. Similar reductions are observed across farms for all three impact categories, although the reductions for terrestrial acidification are slightly higher due to the generally higher contribution of compost production to this impact category (as seen in plots of contributions for individual farms - Figure 26, Figure 33, Figure 40, and Figure 47). The exclusion of composting emissions generally gives similar results as allocation Scenario 2, but results from the latter are slightly lower since Scenario 2 excludes all burdens associated with the composting process for compost inputs (e.g., energy and fuel inputs), not just composting emissions. However, Scenario 2 also includes burdens for composting crop waste on farms, so for some farms Scenario 2 actually results in slightly higher impact levels (e.g., PU-O-3, R-O-1, and R-O-2 for UK kale production).

Thus, the exclusion of composting emissions influences overall results, as was also seen for Scenario 2. In particular, this often results in organic farms having impact levels more similar to each other, since variation between these farms is often spurred by composting impacts. The largest reductions are seen for the urban and peri-urban farm categories, which generally apply the highest amounts of compost. For example, for U.S. kale production, the largest reductions are seen for the urban farm (n=1), with impacts reduced by approximately 40% for global warming and fine particulate matter formation and 80% for terrestrial acidification (Figure 52 and Figure 53). This results in the urban farm category, which usually had the highest impacts in the default Scenario 1, now having impacts similar to the peri-urban and rural organic farm category averages, and in some cases the conventional. This is similarly seen for U.S. tomato production, except that the rural organic farm category generally has higher impacts than the more urban farms in this case (Figure 56 and Figure 57). For UK kale production, the exclusion of composting emissions results in the peri-urban farm category, which had some of the highest impact in Scenario 1, now having global warming and fine particulate matter formation impact levels lower than the conventional farm and similar to the rural organic farm category (which has the lowest impact levels), as seen in Figure 54 and

Figure 55. For terrestrial acidification, impact levels are also significantly lowered and are similar to the conventional farm category and the default impact level of the rural organic farm category (Figure 55). For UK tomato production, although average urban and peri-urban impact levels are reduced (especially the prior), overall trends remain unchanged. The maximum impact reduction is 58%, seen for the urban farm category's terrestrial acidification impacts (Figure 59).

The inclusion of the average carbon sequestration value from compost application also reduces global warming impacts, often to a similar but slightly higher level as the exclusion of composting emissions, as seen in Figure 52, Figure 54, Figure 56, and Figure 58. Percent reductions in organic farm category averages do not exceed 27%. Average global warming potentials for organic farm categories are thus decreased, although generally not enough to change overall trends and rankings. The exception is for UK kale production, where the inclusion of carbon sequestration reduces the peri-urban farm category average to slightly lower than the conventional (Figure 54).

On the other hand, when using the maximum value for carbon sequestration, global warming impact reductions are more severe, ranging anywhere from 10-180% for farm category averages that are affected (see: Figure 52, Figure 54, Figure 56, and Figure 58). This results in global warming impact levels for farms using compost that are generally lower than seen in Scenario 2, but higher than Scenario 3. The highest reductions are again seen for the individual farms that apply the highest levels of compost, as they will have the largest benefits applied in terms of avoided CO_2. As these farms are generally urban and peri-urban farms, these are the farm categories that see the largest changes in average impact levels.

In some cases, global warming potentials for farms and farm category averages become negative when the maximum compost carbon sequestration amount is applied. This changes the trends that have been previously observed between farms and farm categories. In particular, for U.S. production the largest reductions are seen for the urban farm (n=1), with impact levels reduced by approximately 150% for both crops (Figure 52 and Figure 56). This farm has the highest global warming potential for kale production and second highest for tomato production, in comparison to other farm category averages, but is actually seen to have an overall negative global warming potential once maximum carbon sequestration benefits are applied. The U.S. peri-urban farm category also sees reductions in global warming potential for both crop lifecycles, which bring the category average more in line with conventional levels, but do not result in negative impacts.

In the UK, differences are seen for kale and tomato production. For kale production, the only urban farm considered does not use compost, and thus this category does not see any change in impacts. Instead, the greatest reduction (of approximately 150%) is observed for the peri-urban farm category (Figure 54). Impact levels for this category are driven in particular by two farms using relatively high levels of compost (PU-O-1 and PU-O-3). The application of maximum carbon sequestration benefits results in three out of four peri-urban farms having overall negative global warming potentials, shifting the category average to a negative value that is now the lowest among all farm categories. Finally, for UK tomato production, the inclusion of the maximum level of carbon sequestration reduces both urban and peri-urban farm category averages by 150-180% (Figure 58). This results these farm category averages being at similar levels that are also negative and now the lowest out of all farm categories.

Thus, it can be seen that the exclusion of composting emissions and inclusion of average carbon sequestration benefits does lower impacts for farms, but generally not enough to change the trends observed between farm categories. The inclusion of the maximum carbon sequestration value can generate negative overall global warming potentials that does shift results, generally seen as the urban and peri-urban farm categories become those with the lowest global warming impacts. However, there is a high degree of uncertainty around this value, which has been included here mainly for the benefit of comparing with other values used in literature. Overall, the attribution of avoided burdens from the municipal solid waste stream in Scenario 3 still results in the lowest global warming, fine particulate matter formation, and terrestrial acidification impacts for farms using compost.

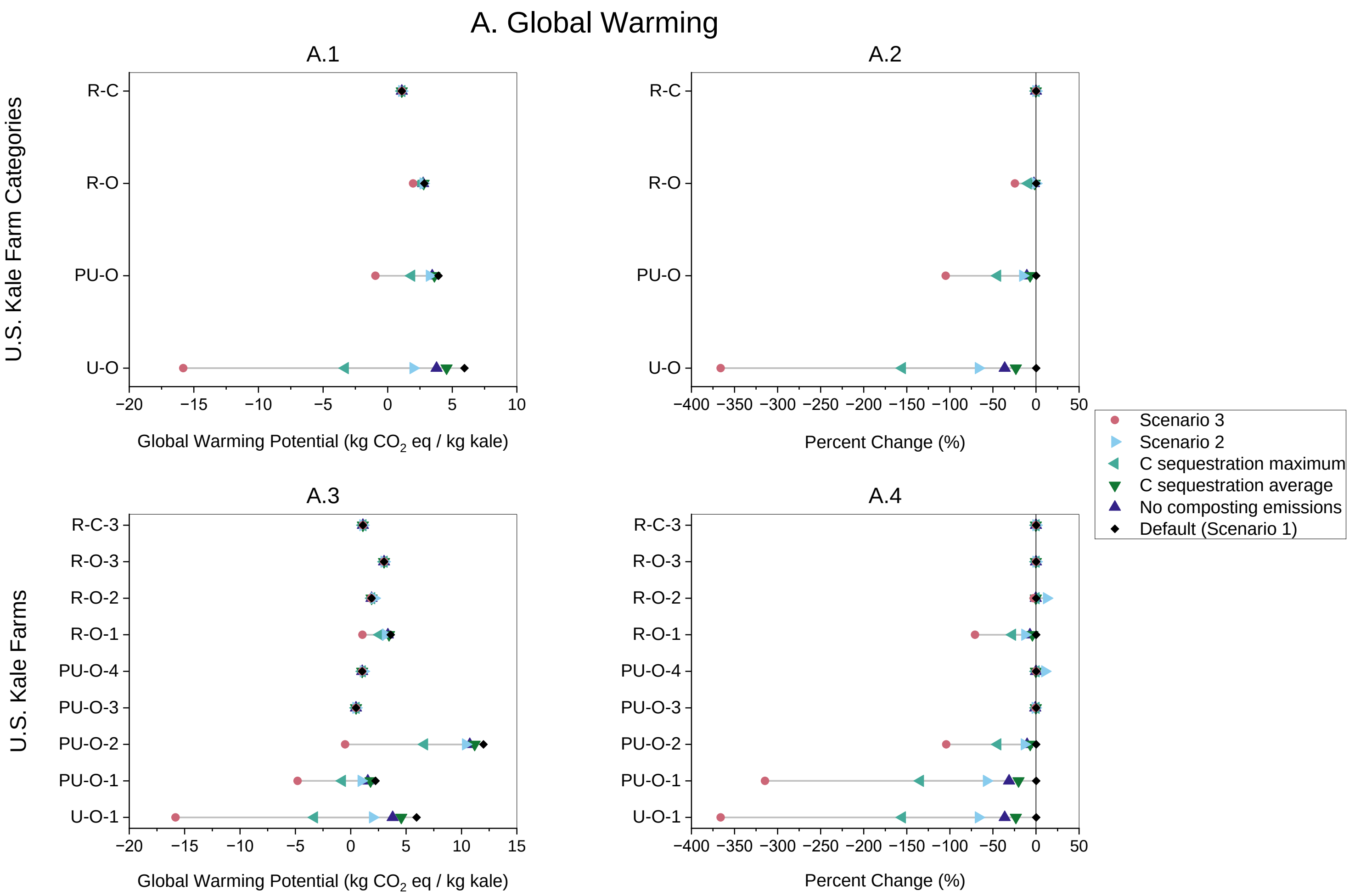

***Figure 52** – Global warming potential impacts (A) from the compost sensitivity analysis for U.S. kale production. Plots display default impact levels from Scenario 1 and the impact levels generated by each sensitivity case, which are: the exclusion of composting emissions and inclusion of carbon sequestration benefits (as designated in the legend). Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (A.1) and for individual farms (A.3), per kg kale. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (A.2) and individual farms (A.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

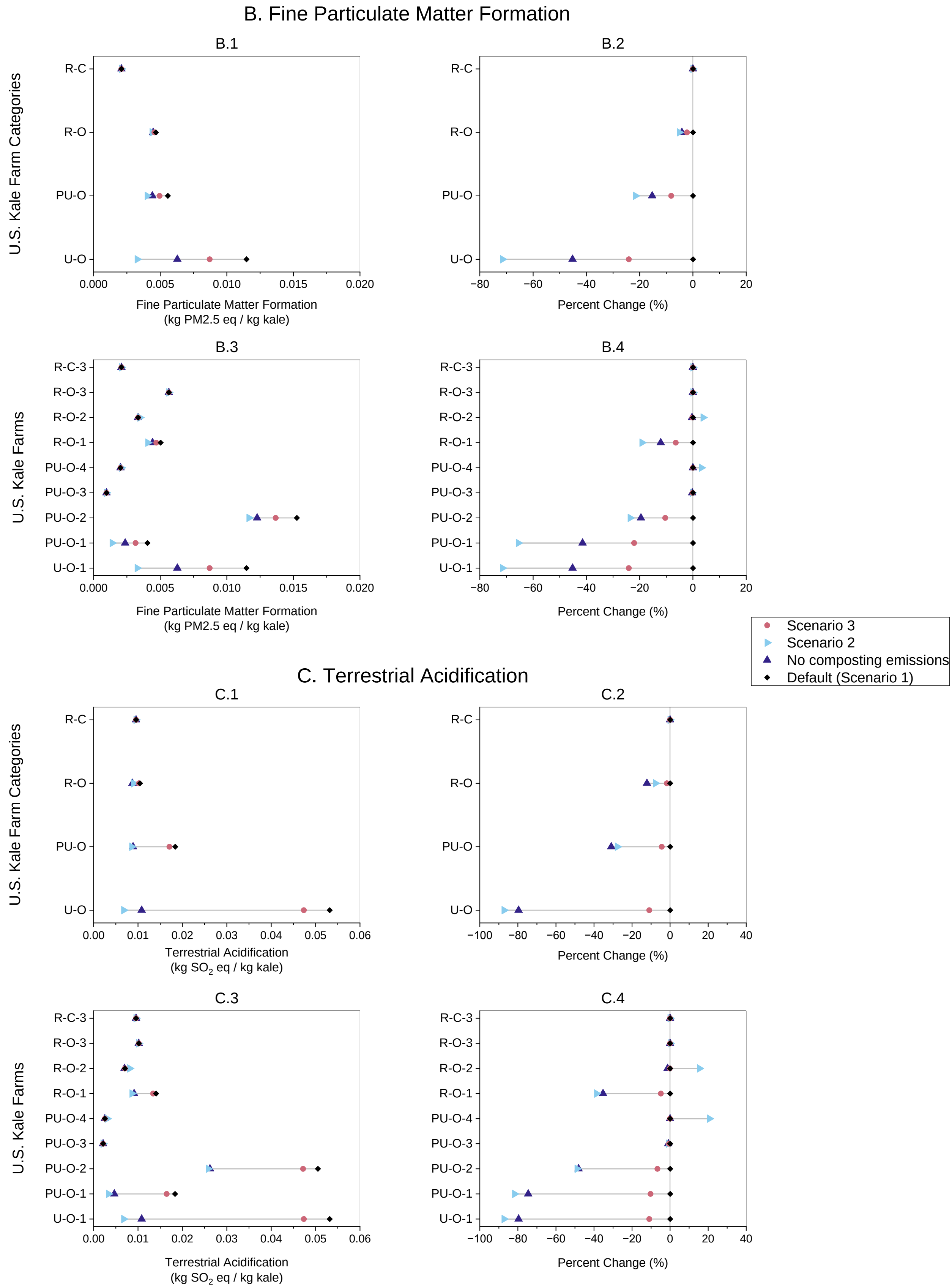

Figure 53 *– Fine particulate matter formation (B) and terrestrial acidification (C) impacts from the compost sensitivity analysis for U.S. kale production. Plots display default impact levels from Scenario 1 and those generated by testing the exclusion of composting emissions. Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (B.1, C.1) and for individual farms (B.3, C.3), per kg kale. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (B.2, C.2) and individual farms (B.4, C.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

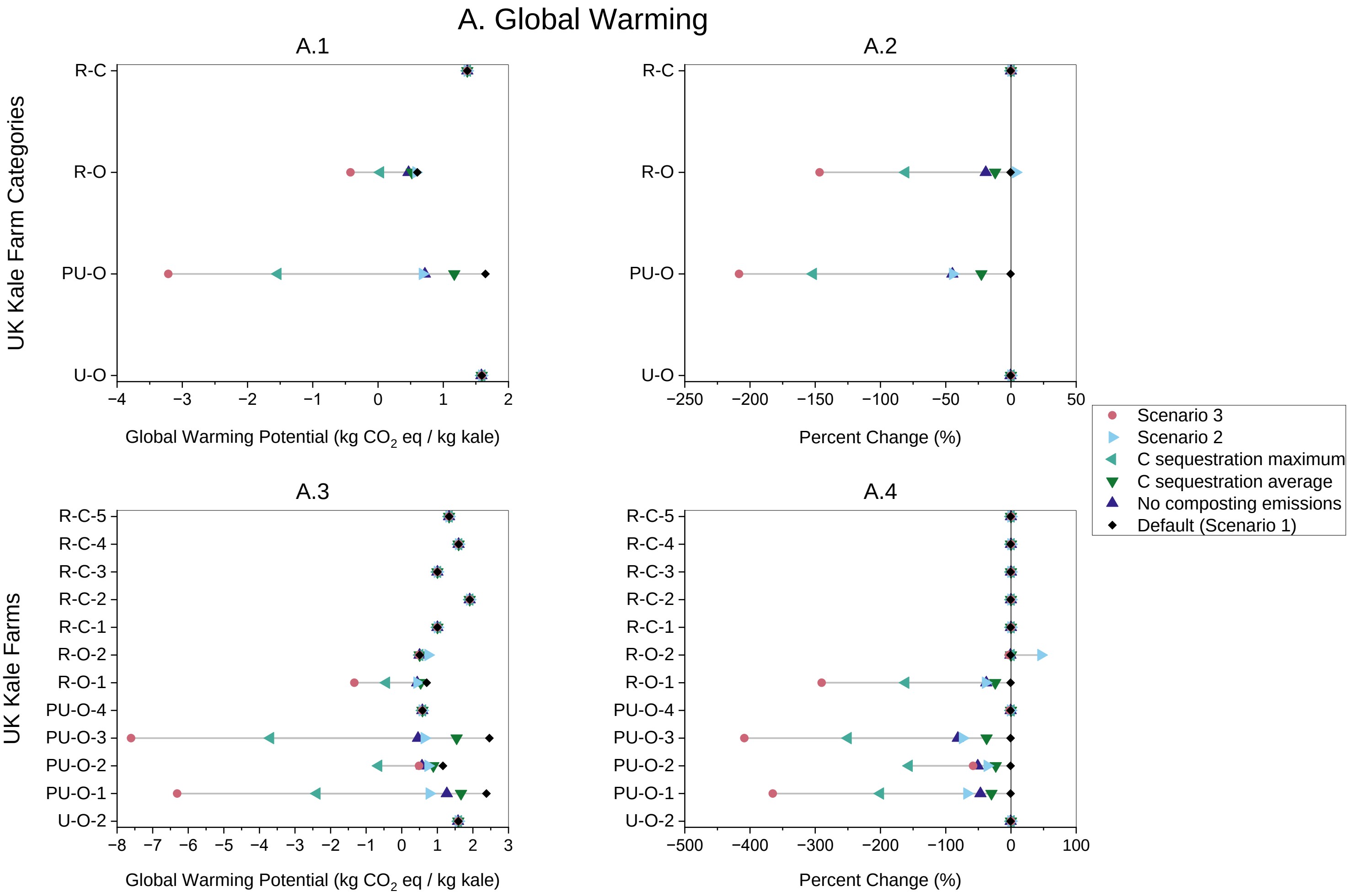

Figure 54 *– Global warming potential impacts (A) from the compost sensitivity analysis for UK kale production. Plots display default impact levels from Scenario 1 and the impact levels generated by each sensitivity case, which are: the exclusion of composting emissions and inclusion of carbon sequestration benefits (as designated in the legend). Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (A.1) and for individual farms (A.3), per kg kale. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (A.2) and individual farms (A.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

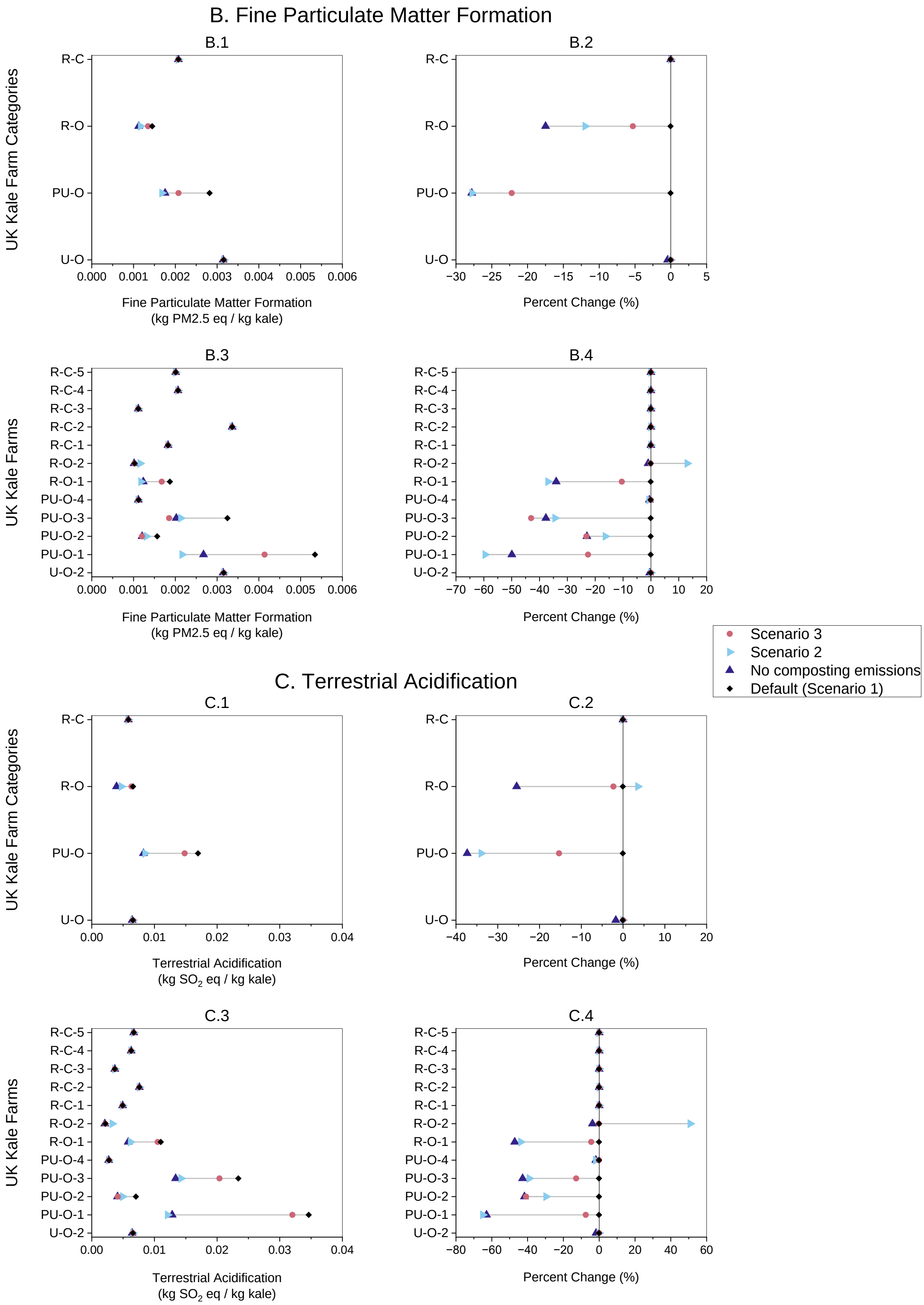

***Figure 55** – Fine particulate matter formation (B) and terrestrial acidification (C) impacts from the compost sensitivity analysis for UK kale production. Plots display default impact levels from Scenario 1 and those generated by testing the exclusion of composting emissions. Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (B.1, C.1) and for individual farms (B.3, C.3), per kg kale. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (B.2, C.2) and individual farms (B.4, C.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

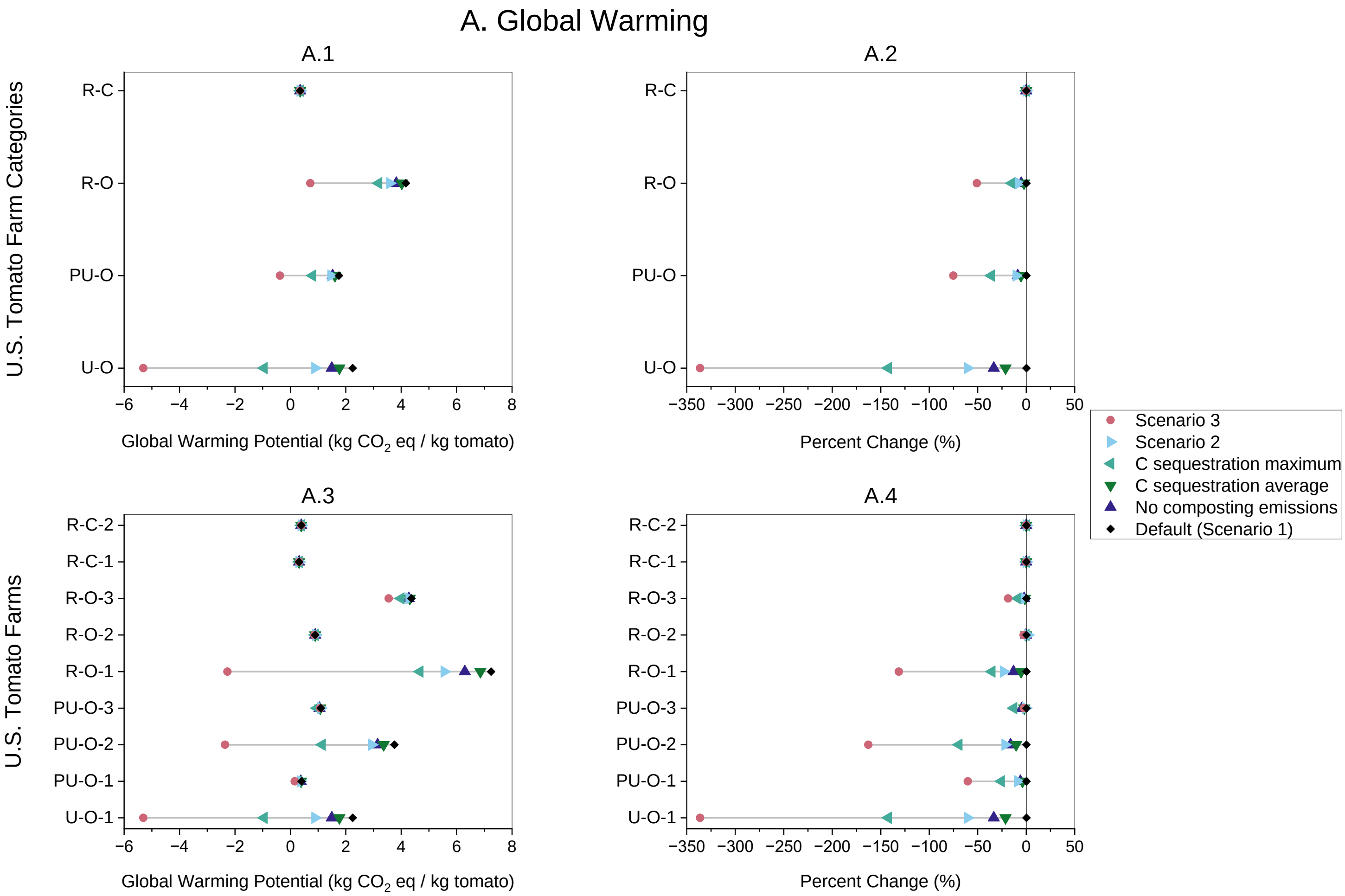

***Figure 56** – Global warming potential impacts (A) from the compost sensitivity analysis for U.S. tomato production. Plots display default impact levels from Scenario 1 and the impact levels generated by each sensitivity case, which are: the exclusion of composting emissions and inclusion of carbon sequestration benefits (as designated in the legend). Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (A.1) and for individual farms (A.3), per kg tomato. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (A.2) and individual farms (A.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

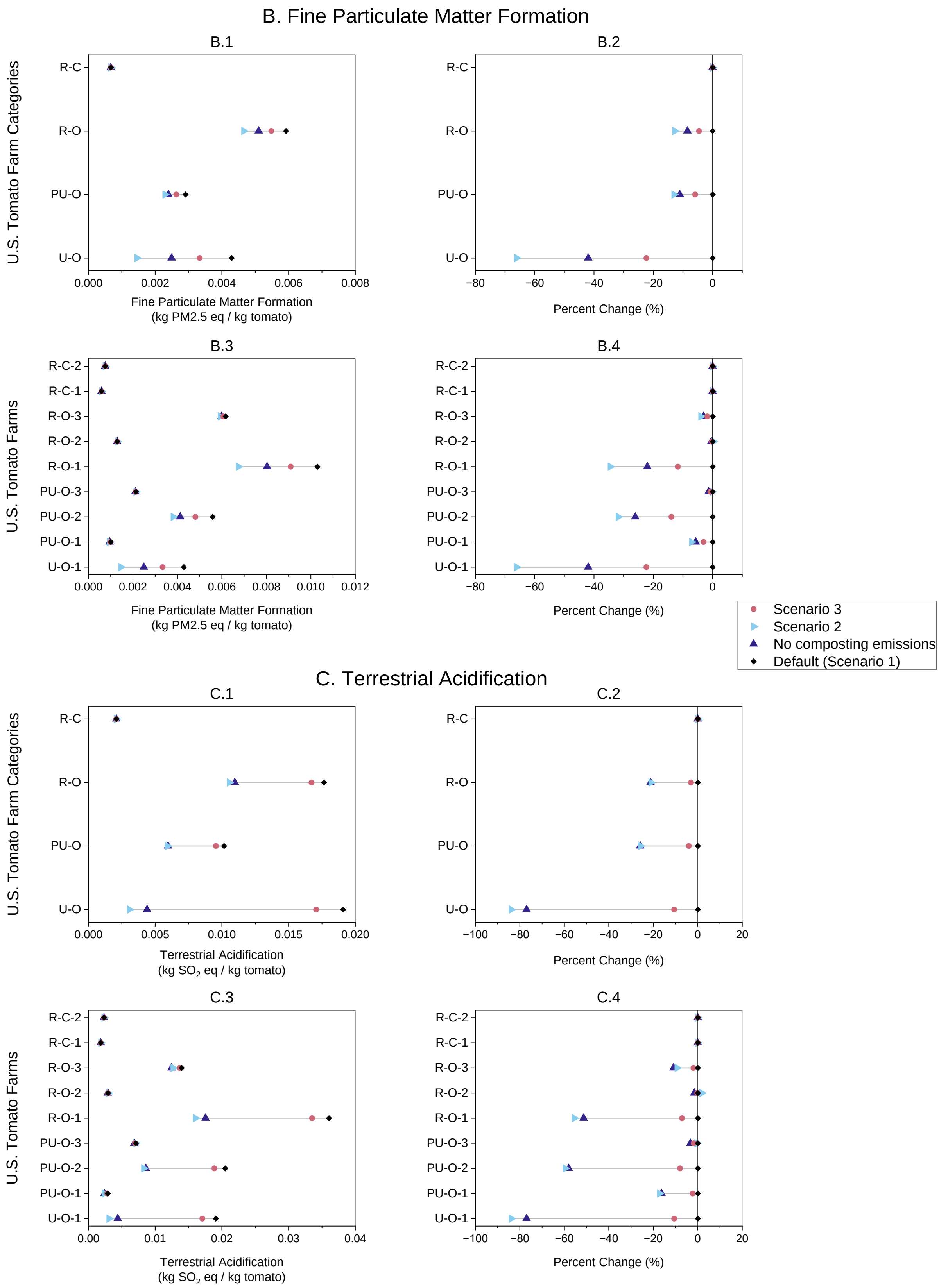

Figure 57 *– Fine particulate matter formation (B) and terrestrial acidification (C) impacts from the compost sensitivity analysis for U.S. tomato production. Plots display default impact levels from Scenario 1 and those generated by testing the exclusion of composting emissions. Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (B.1, C.1) and for individual farms (B.3, C.3), per kg tomato. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (B.2, C.2) and individual farms (B.4, C.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

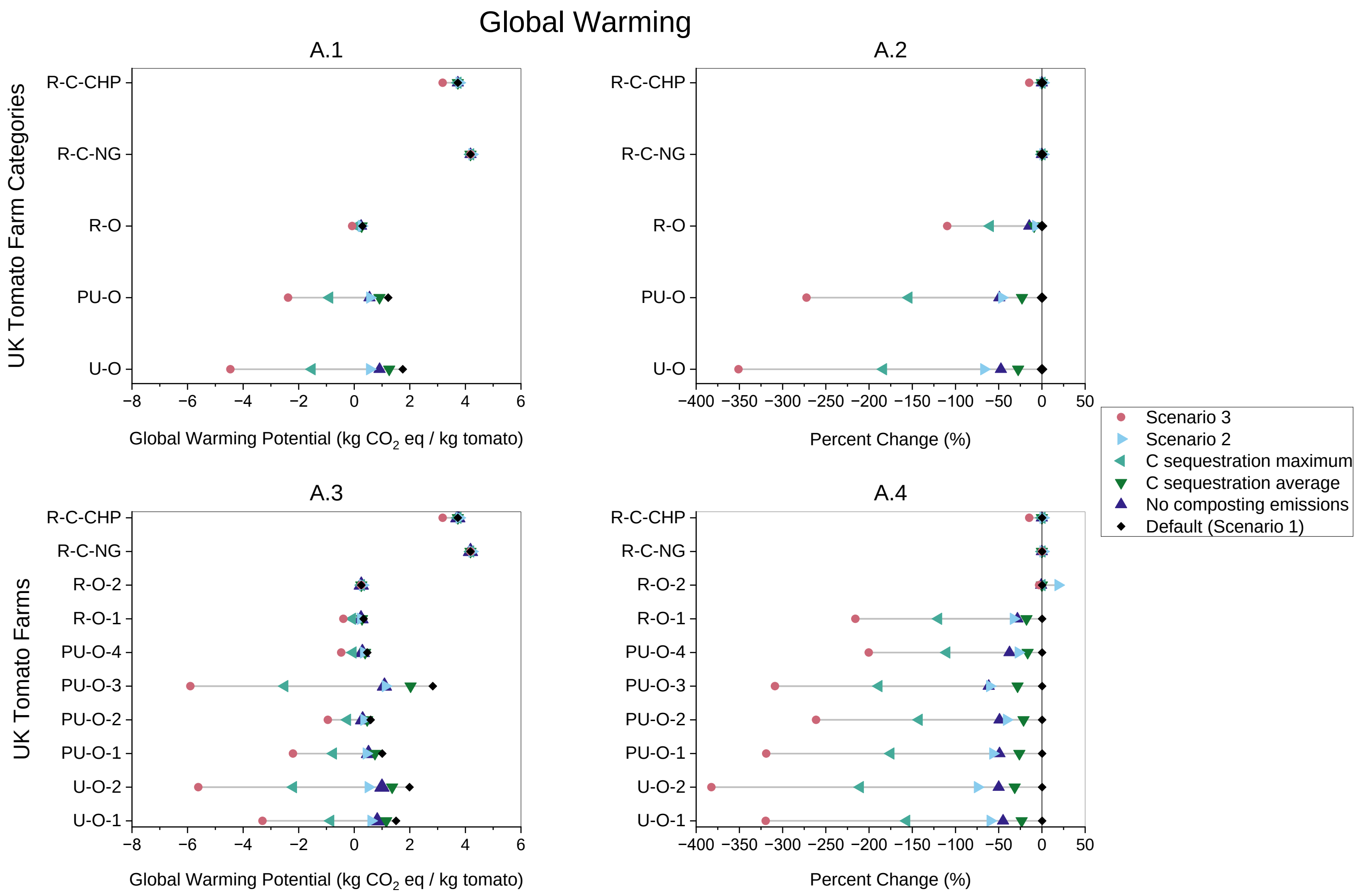

Figure 58 *– Global warming potential impacts (A) from the compost sensitivity analysis for UK tomato production. Plots display default impact levels from Scenario 1 and the impact levels generated by each sensitivity case, which are: the exclusion of composting emissions and inclusion of carbon sequestration benefits (as designated in the legend). Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (A.1) and for individual farms (A.3), per kg tomato. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (A.2) and individual farms (A.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

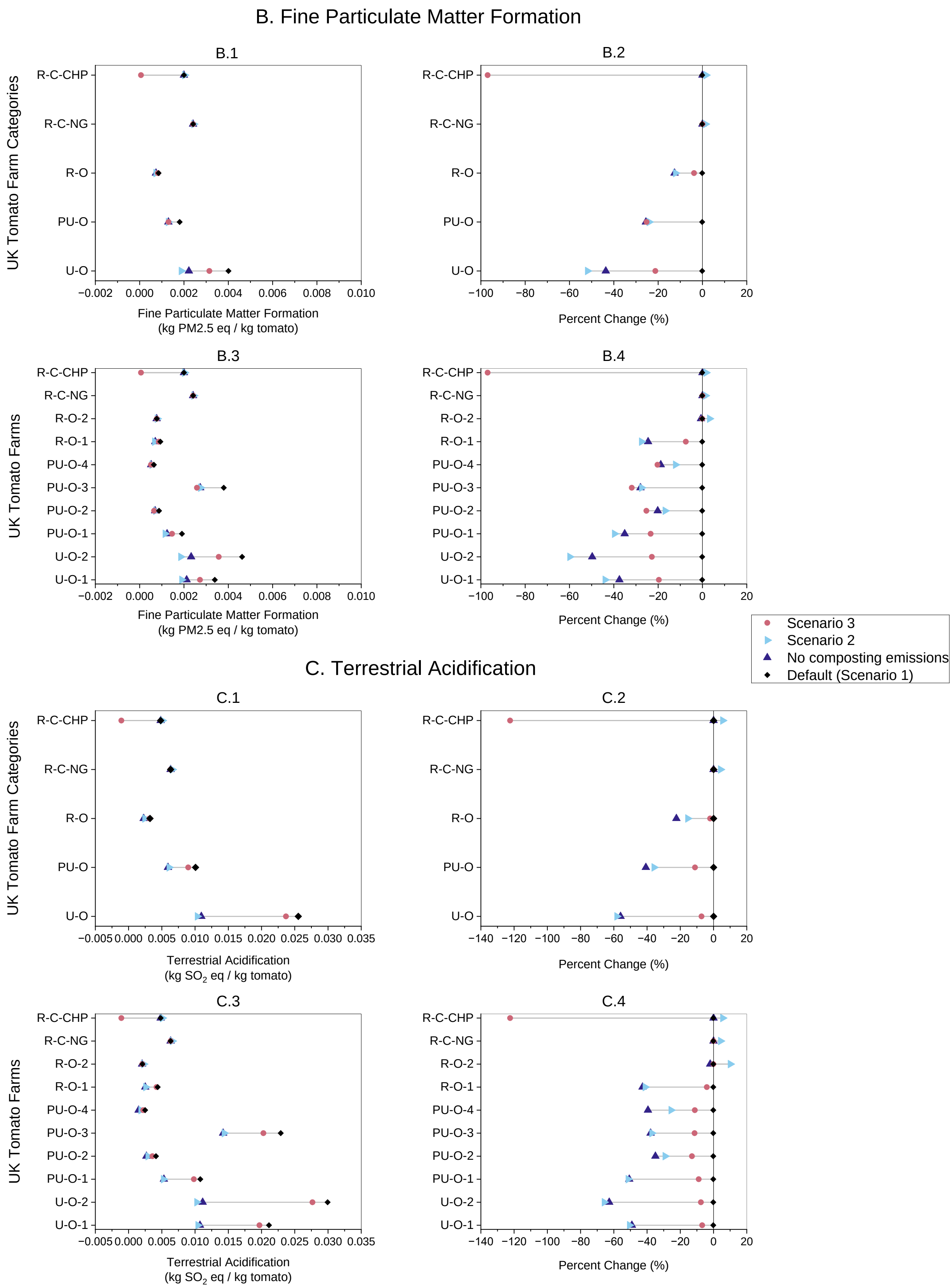

Figure 59 – *Fine particulate matter formation (B) and terrestrial acidification (C) impacts from the compost sensitivity analysis for UK tomato production. Plots display default impact levels from Scenario 1 and those generated by testing the exclusion of composting emissions. Results from Scenario 2 and Scenario 3 are provided for comparison purposes. Actual impact levels are displayed as farm category averages (B.1, C.1) and for individual farms (B.3, C.3), per kg tomato. Plots depicting the percent change in impact levels from the default for each sensitivity case are presented next to those with actual values, for farm category averages (B.2, C.2) and individual farms (B.4, C.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

3.1.3.2 Nitrate leaching

Nitrate leaching emissions from agricultural soils are the main factor in determining marine eutrophication impacts within this LCA (see: Figure 28, Figure 35, Figure 42, and Figure 49). The choice of nitrate leaching model has also been found to be a source of uncertainty and dissonance throughout LCA literature (Henryson *et al.*, 2020; Avadí *et al.*, 2022). Thus, for this study, results were tested using five different nitrate leaching models to understand if and how the choice of model influences final results.

The default model, as used to generate all prior results, is the de Willigen 2000 model (de Willigen, 2000). The other four models that have been tested within this sensitivity analysis include: the SQCB-NO_3 model (Emmenegger, Reinhard and Zah, 2009); the Smaling 1993 model (Smaling, Stoorvogel and Windmeijer, 1993); the Poore-Nemecek model (Poore and Nemecek, 2018b); and the IPCC 2019 Tier 1 emission factor for nitrate leaching (IPCC, 2019). The prior three models, as well as the default model, all require some site-specific data for nitrate leaching calculations, although to varying degrees of detail. The default model and the SQCB-NO_3 model require the most rigorous and specific site information (e.g., clay content of soil, precipitation and irrigation levels, rooting depths, and calculations of mineralizable N). These models also the only ones to consider N flux (e.g., N input to soil vs. N uptake from crop biomass). The Smaling and Poore-Nemecek model also require site-specific information, but not necessarily to perform calculations; for example, in the prior, clay content is used to determine the calculation equation used, which takes into account N inputs, precipitation, and irrigation; for the latter, environmental conditions are used to determine low, medium, and high leaching risks, for which different emission factors are defined. On the other hand, the IPCC 2019 Tier 1 emission factor is site generic and thus is applied solely to amounts of N input. It should be noted that the assumption that nitrate leaching is zero for drip-irrigated, indoor production systems (per IPCC 2019 guidelines), which is applied throughout the default model, is also held throughout all other leaching models for consistency. Section 2.1.9.5.2 provides further detail on all nitrate leaching models and methods of calculation.

An additional consideration has also been included within the default nitrate leaching model to calculate the N inputs based on soluble N only, instead of total N. This is particularly important for farms using organic fertilisers (e.g., compost and manure), as generally the proportion of soluble N is much less than total N, with these fertilisers generally assumed to be slow releasing. On the other hand, for synthetic and mineral fertilisers, it is assumed that all N is soluble. This is consistent with assumptions used for nitrate leaching within the WFLDB v.3.0 database, where total N is considered for all mineral fertilisers and only the soluble portion of fertiliser N is considered for organic fertilisers (Nemecek *et al.*, 2019). This is only tested within the default model; all other nitrate leaching models as included within this sensitivity analysis are still calculated using total N.

The nitrate leaching models as previously discussed are only valid for soil-based systems. Thus, for UK conventional tomato production (R-C-NG and R-C-CHP), which operates through soilless systems, nitrate leaching has been estimated using an emission factor averaged from literature (assuming 45% NO_3-N leached per applied N; see Section 2.1.9.5.3 for more detail). It is assumed that leaching only occurs through open (free-draining) hydroponic systems, which constitute 50% of the systems used by the farm in this study. This assumption has been tested within this sensitivity analysis by also applying the nitrate leaching emission factor to closed (recirculating) hydroponic systems, thus assuming that leaching occurs from all hydroponic systems. The results from this have been presented for UK tomato production alongside the analysis of the other nitrate leaching models for soil-based cultivation, but are also included again within the other sensitivity analysis results for soilless system emissions.

The selection of nitrate leaching model mainly affects the marine eutrophication category, but also influences global warming potential and stratospheric ozone depletion due to the indirect N_2O emissions generated from leached nitrate (see: Section 2.1.9.6.4). Changes in impacts based on nitrate leaching model have been presented for the prior two impact categories only, as these are within the seven focus impact categories for this LCA.

The influence of nitrate leaching models on global warming and marine eutrophication impacts are presented U.S. kale production (Figure 60), UK kale production (Figure 61), U.S. tomato production (Figure 62), and UK tomato production (Figure 63). Throughout these results, it can be seen that the choice of nitrate leaching model does not significantly influence global warming results but does influence results for marine eutrophication. For global warming potential, the switch of nitrate leaching models results in a maximum percent change from default values on individual farms of: 3% for U.S. kale production; 11% for UK kale production; 6% for U.S. tomato production; and 2% for UK tomato production. For UK conventional production, the assumption of leaching occurring for all hydroponic systems barely changes global warming potential results, showing a percent change of <0.08%. Thus, basic trends for global warming potential as observed between farm categories and individual farms remain unchanged between nitrate leaching models.

For marine eutrophication, changes in impact are more severe and can influence the overall trends observed. In particular, changes of up to 4,700% in impact level are seen, although generally changes are <200%. The largest changes in marine eutrophication impact levels based on nitrate leaching model are generally observed for the urban and / or peri-urban farm categories. For example, increases of >200% in marine eutrophication levels are seen for US-PU-O-1 for kale production and US-PU-O-2 for both tomato and kale production. This is likely due to the much higher levels of compost application on these farms, and thus higher levels of total applied N which become more sensitive within leaching calculations, especially for the IPCC calculation that just applies an emission factor to total applied N.

The model that generates the highest or lowest impact values varies depending on the particular farm. Generally, the default model (de Willigen) appears to generate either the second highest or highest impact levels in comparison to the other nitrate leaching models. However, results from the SQCB and de Willigen model are often similar, especially for the U.S. crop lifecycles, which is sensible as the SQCB model is an adaptation of the original de Willigen equations. For UK crop lifecycles, however, the SQCB model tends to give results more similar to the Poore-Nemecek model.

The other model generating relatively high impact levels is the IPCC (2019) model, as this is just an emission factor that is equally applied to total N input values across all farms, regardless of any specific environmental conditions. The IPCC emission factor will thus lead to particularly high leaching values for farms applying higher amounts of N (per kg of produced crop) or farms that have moderate N inputs but with high yields, and thus may have seen lower leaching values in the default model because it considers N flux (i.e., accounts for N exported in crop biomass). For example, this is seen for US-PU-O-1 and UK-R-O-1 for kale production. However, the IPCC model may also lead to values lower than the default, especially for farms that apply moderate or low amounts of total N, but may have had higher leaching values in other models due to being in a geographical area with higher risk leaching conditions. This is especially seen within UK kale production, as this takes place outdoors for almost every farm and the UK has generally higher precipitation values than seen in Georgia. In particular, U-O-2 and all conventional farms (except R-C-5) have the highest marine eutrophication impacts from the default nitrate leaching model, due to the relatively high precipitation levels seen for these farms (see: Table 55). However, in general, if the IPCC model was considered across all lifecycles, this would result in all organic farm categories having marine eutrophication impacts either similar to or higher than conventional farm averages.

On the other hand, the Poore-Nemecek model tends to result in the lowest impact levels, seen especially for conventional farms across both crops and countries (excluding UK tomato production). This model also generates relatively low impact levels for organic farms, usually seen as the second lowest impact level after the those generated by considering only soluble N within the default model (de Willigen). The inclusion of soluble N mainly affects organic farms, which tend to apply organic inputs with relatively low amounts of soluble N, in comparison to organic farms, where generally most or all applied N in mineral fertilisers is soluble. Thus, if the default model was applied considering only soluble N, this would make average marine eutrophication impacts for all organic farm categories lower than or similar to respective conventional averages. This changes average trends for all crop and country lifecycles. In addition, for U.S. kale and UK tomato production, using the default model with only soluble N would actually make each organic farm impact level (individually) lower than the conventional average for marine eutrophication. Since soluble N is considered only as a sensitivity within the default nitrate leaching model, which often results in some of the highest impact levels normally, then it can be presumed that including only soluble N within other leaching models would result in even lower impacts. This shows the importance of considering the susceptibility of different fertilisers to leaching within modelling efforts.

Although in most cases the urban and peri-urban farm categories have the highest changes in impact levels, in UK kale production the rural farm category average sees drastic changes in impact levels. In particular, the switch to the Poore-Nemecek model results in an approximate 630% increase in marine eutrophication impact whilst the switch to the IPCC (2019) models results in an approximate 2,300% increase in impact for the rural organic farm category (n=2). However, it can be seen that this drastic change is driven primarily by one farm (R-O-1). This farm sees such a drastic increase in impact level (e.g., 4,700% when switching to the IPCC model) because the default model generated a leaching value of zero. This is due the N flux (N applied minus N exported in crop biomass) being negative for this farm, driven in part by the relatively high yields seen on this farm compared to other organic farms. Thus, when other models are applied that do not consider N flux (which are all models except the default and SQCB), this results in a large increase in the marine eutrophication level. However, in real terms, the impact levels generated from other nitrate leaching models for R-O-1 are similar to (or even still lower) than the other organic farms. Excluding R-O-1, all other UK kale farms experienced changes in marine eutrophication impact levels of no more than 100% based on leaching model.

There are also specific considerations for UK tomato production. For one, the majority of results are unchanged as most organic farms grew tomatoes completely indoors using drip irrigation, and thus are assumed to have nitrate leaching values of zero. Nitrate leaching is non-zero for only UK-U-O-2, UK-PU-O-3, and UK-R-O-2, out of the organic farms. For these farms, the lowest impact levels are similarly seen when either considering only soluble N within the default model or when applying the Smaling (1993) model. This is in contrast to the other crop and country lifecycles, where generally the Poore-Nemecek model generated lower values for organic farms. For all models except for the Smaling and the consideration of soluble N, U-O-1 and PU-O-3 show relatively high marine eutrophication impacts compared to other farms, whilst the impact level for R-O-2 is near-zero and thus similar to the other organic farms for which nitrate leaching is assumed to be zero. This is because of the relatively high amounts of compost applied on both U-O-2 and PU-O-3. Finally, for UK conventional (soilless) production, it can be seen that the inclusion of leaching estimates from closed systems generates an increase in of marine eutrophication impacts of approximately 27%. In real terms, this does not significantly change trends or results.

Overall, it can therefore be seen that the selection of nitrate leaching model does influence overall results and conclusions for the marine eutrophication category, but does not significantly affect global warming potentials within this LCA.

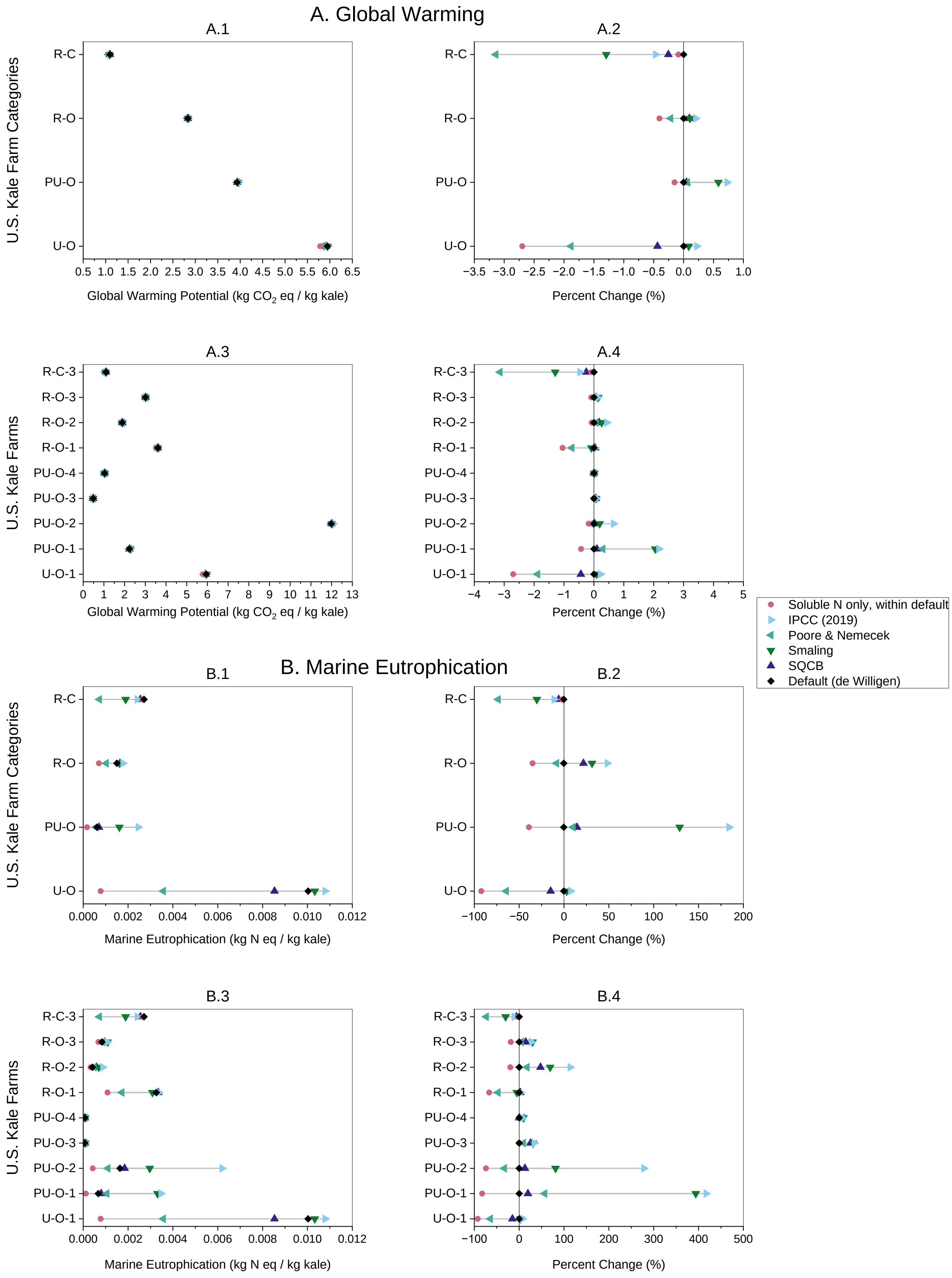

Figure 60 *– Global warming (A) and marine eutrophication (B) impacts generated from different nitrate leaching models for U.S. kale production. Actual impact levels are displayed as farm category averages (A.1, B.1) and for individual farms (A.3, B.3) , per kg kale. Plots depicting the percent change in impact levels from the default for each nitrate leaching model are presented to next to those with actual values, for farm category averages (A.2, B.2) and individual farms (A.4, B.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

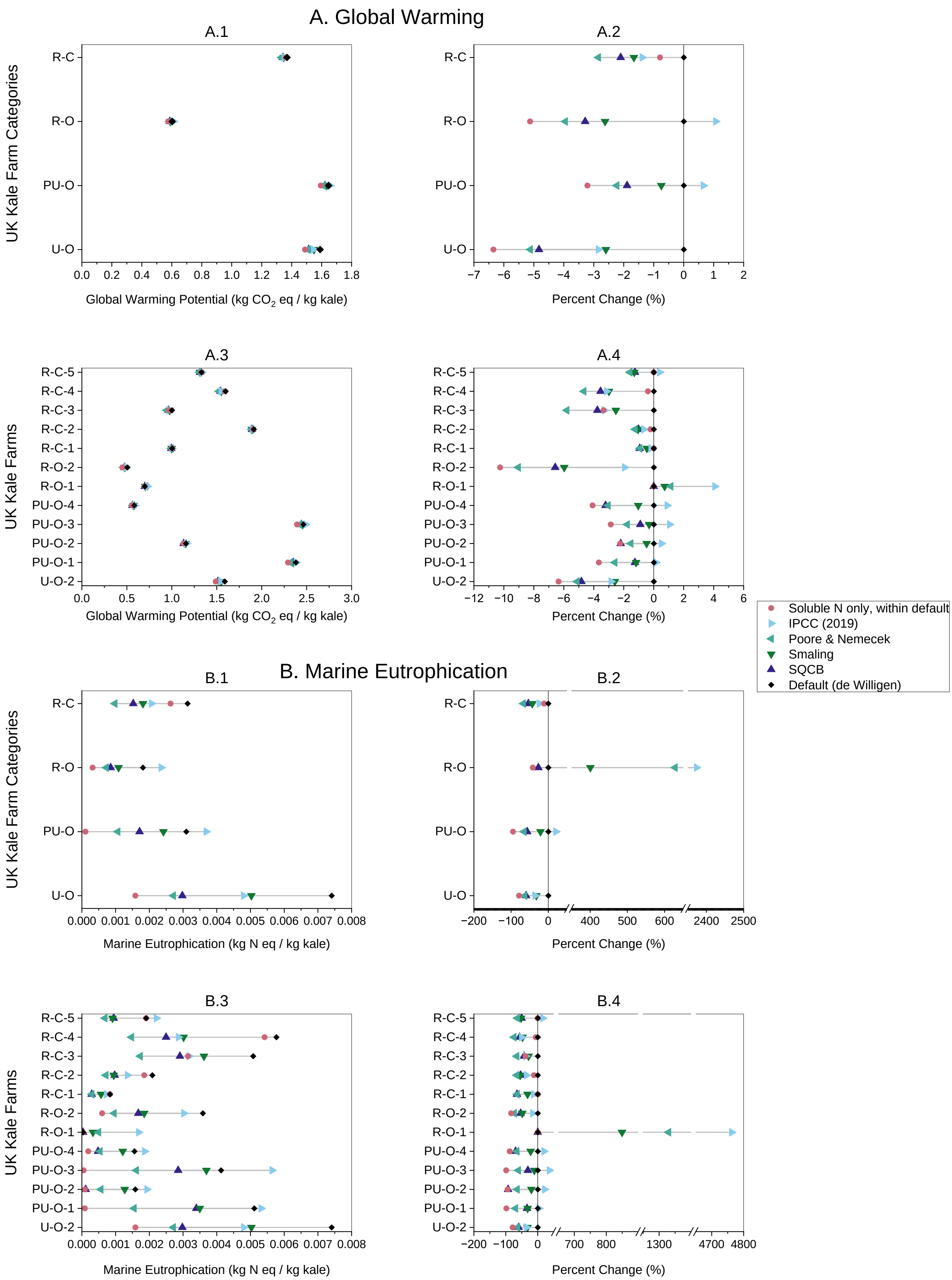

Figure 61 *– Global warming (A) and marine eutrophication (B) impacts generated from different nitrate leaching models for UK kale production. Actual impact levels are displayed as farm category averages (A.1, B.1) and for individual farms (A.3, B.3), per kg kale. Plots depicting the percent change in impact levels from the default for each nitrate leaching model are presented to next to those with actual values, for farm category averages (A.2, B.2) and individual farms (A.4, B.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value. Note that breaks in the axis are included within percent change plots for marine eutrophication.*

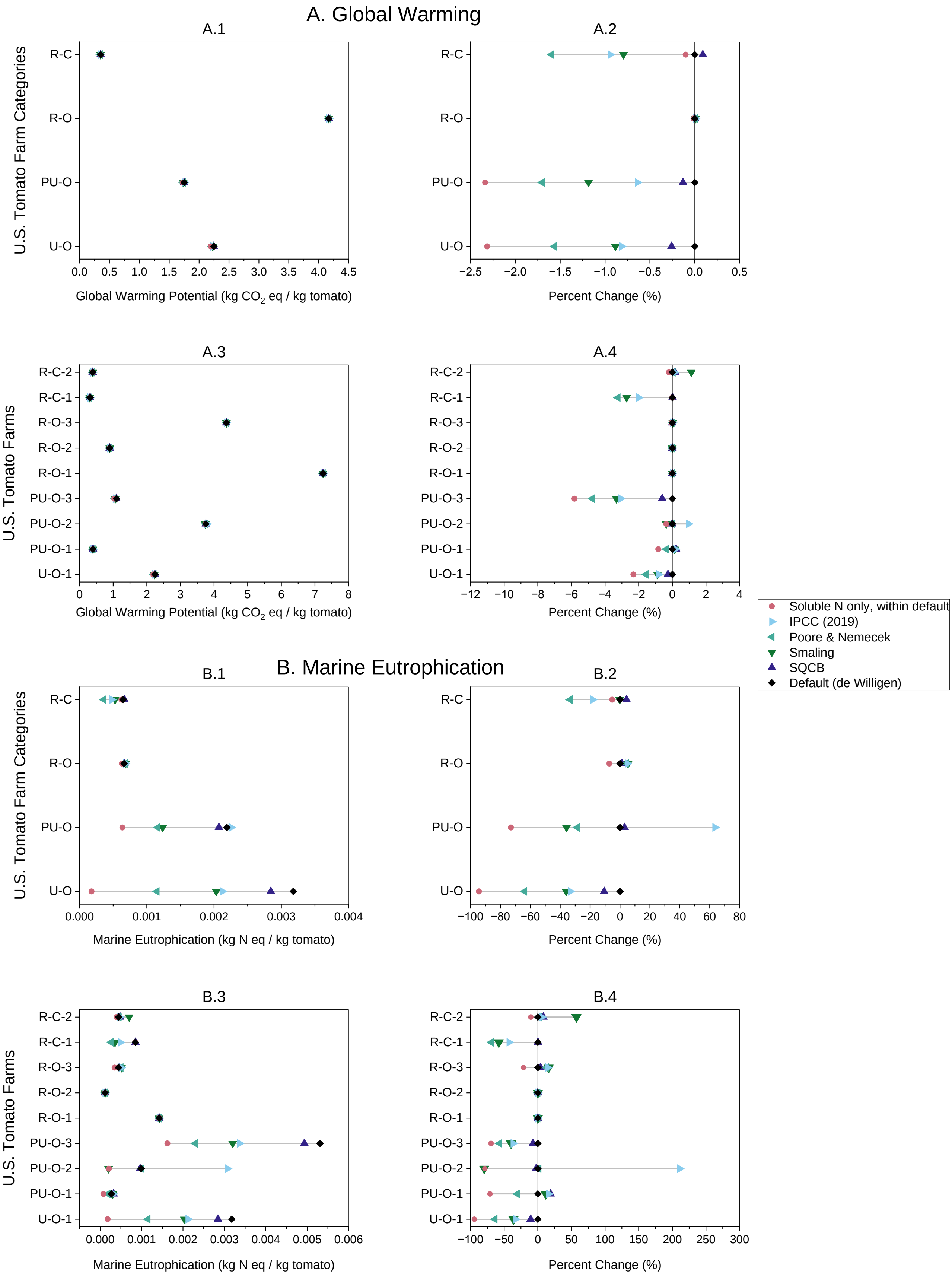

***Figure 62** – Global warming (A) and marine eutrophication (B) impacts generated from different nitrate leaching models for U.S. tomato production. Actual impact levels are displayed as farm category averages (A.1, B.1) and for individual farms (A.3, B.3), per kg kale. Plots depicting the percent change in impact levels from the default for each nitrate leaching model are presented to next to those with actual values, for farm category averages (A.2, B.2) and individual farms (A.4, B.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

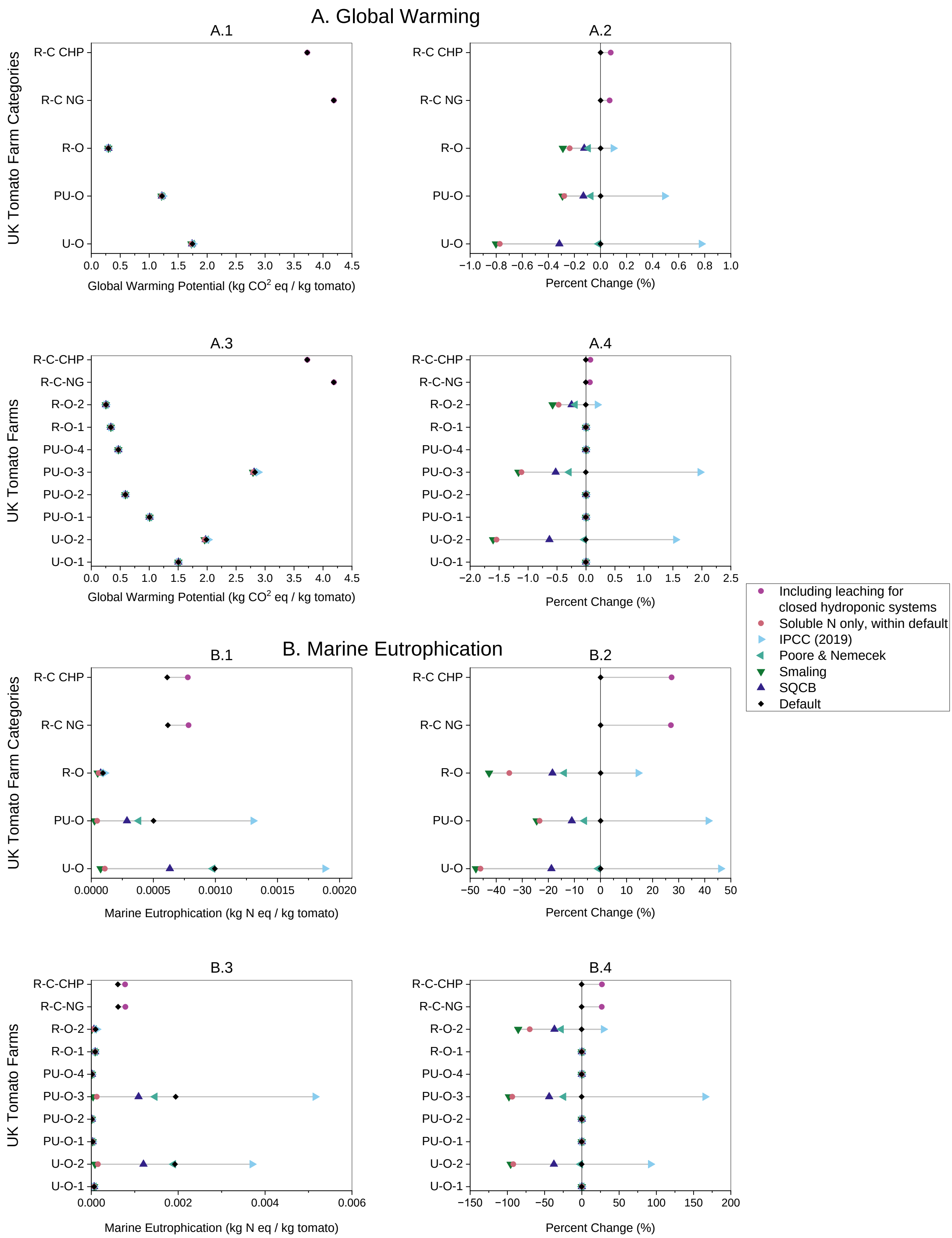

Figure 63 *– Global warming (A) and marine eutrophication (B) impacts generated from different nitrate leaching models for UK tomato production. Actual impact levels are displayed as farm category averages (A.1, B.1) and for individual farms (A.3, B.3), per kg kale. Plots depicting the percent change in impact levels from the default for each nitrate leaching model are presented to next to those with actual values, for farm category averages (A.2, B.2) and individual farms (A.4, B.4). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

3.1.3.3 Soilless systems

The emissions generated during soilless crop cultivation have been a source of contention and dissonance across LCA studies. Thus, soilless emission levels have been tested in this LCA to see how sensitive the results for UK conventional tomato production are to the emission assumptions made in this LCA. N_2O is one of the most important substances driving global warming potential across farms in this study (Table 62), and thus, results have been re-generated for UK conventional tomato production using minimum and maximum N_2O levels that have been actually measured from soilless systems, as found in literature (Daum and Schenk, 1996a; Yoshihara *et al.*, 2014; Llorach-Massana *et al.*, 2017; Karlowsky *et al.*, 2021). The default value used throughout this study is 0.0087 kg N_2O-N (kg applied N)$^{-1}$, generated as an average from a variety of literature sources. The minimum value tested is 0.001 kg N_2O-N (kg applied N)$^{-1}$, as measured in a study by Karlowsky *et al.* (2021), and the maximum value is 0.046 kg N_2O-N (kg applied N)$^{-1}$, as measured in a study by Yoshihara *et al.* (2014). This sensitivity analysis only affects the global warming potential and stratospheric ozone depletion categories, with the prior presented within these results.

Additionally, some LCA studies do not include N_2O or other N-related air emissions from fertiliser application at all, because of the uncertainty around whether these emissions do in fact occur (Almeida *et al.*, 2014; Dias *et al.*, 2017). Thus, an additional sensitivity analysis has been performed by excluding all N-related air emissions from fertiliser use; this includes N_2O, NH_3 emissions, and NO_2 emissions. The latter two emissions were estimated in the original results using EEA 2019 guidance (EMEP & EEA, 2019a), the same methodology used for soil-based cultivation. More detail on the default methods for determining these emissions are provided in Section 2.1.9.6.2, 2.1.9.4, and 2.1.9.3.1, for N_2O, NO_2, and NH_3, respectively. The exclusion of these emissions affects the global warming, fine particulate matter formation, terrestrial acidification, stratospheric ozone depletion, and ozone formation categories, with results presented for the prior three as impact categories of interest.

There is also some uncertainty around nitrate emissions to water from leaching in soilless systems. Originally, this has been estimated by assuming 0.45 kg NO_3^- is leached per kg N input, as derived from an average of studies that measured nitrate contents in leachate from open hydroponic systems growing tomatoes (Massa *et al.*, 2010; Thompson *et al.*, 2013; Sanjuan-Delmás *et al.*, 2020). This value was only applied to the portion of open (free-draining) hydroponic systems used by the UK conventional tomato farm, as nitrate emissions occur from the leachate, and there is assumed to be low or no leachate from closed (recirculating) hydroponic systems. However, this has been tested by also applying the nitrate leaching value to the portion of closed hydroponic systems employed by the farm (50% of systems), to see how this assumption influences final results. This is relevant to global warming, freshwater eutrophication and marine eutrophication impact categories, freshwater and marine ecotoxicity, human non-carcinogenic toxicity, and stratospheric ozone depletion categories, with results for the prior three presented as within the impact categories of interest.

Finally, the last analysis conducted on soilless systems was to test how the proportion of open and closed hydroponic systems influences final results. Thus, the results were re-generated assuming all open hydroponic systems and all closed hydroponic systems, with the default value being 50/50. Although the 50/50 value was provided by the grower, and thus represents the real case on the farm, the proportion of open and closed systems was tested to provide insight to how these different hydroponic systems might influence environmental impacts and to evaluate the possible advantage or disadvantages of one system over another. The assumption of closed vs. open hydroponic systems in particular changes levels of fertiliser inputs, water inputs, and nitrate leaching emissions. Closed hydroponic systems are assumed to use 30% less water and 50% less fertiliser throughout this LCA (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018). Within this sensitivity analysis, closed hydroponic systems are also still assumed to have zero leaching, as in the default methods. Thus, the assumption of 100% open or 100% closed hydroponic systems influences all impact categories, so results from all seven impact categories of interest have been provided (i.e., global warming, fine particulate matter formation, terrestrial acidification, freshwater eutrophication, marine eutrophication, water consumption, and land use).

Results from these sensitivity analyses have been collectively presented for all impact categories. Results are presented only for UK tomato production, as this is the only country and crop lifecycle that includes soilless production. Results are shown for both R-C-NG, the case of soilless cultivation using conventional heating and electricity, and R-C-CHP, the case of soilless production using combined heat and power. These have been presented alongside other UK tomato farm category averages for reference and comparison purposes, but these farm category averages remain unchanged throughout these sensitivity analyses as the tested assumptions do not affect soil-based cultivation.

Figure 64 displays results for global warming (A), fine particulate matter formation (B), and terrestrial acidification (C), all of which are influenced by the exclusion of air emissions and the assumption of 100% open and 100% closed hydroponic systems. Global warming potential is also influenced by use of the minimum and maximum N_2O emission factor for direct N_2O emissions and by the inclusion of nitrate leaching emissions for closed hydroponic systems, because this also results in indirect N_2O emissions. Overall, it can be seen that none of the tested sensitivities lead to any significant change in results for these impact categories. Even when employing the maximum N_2O emission factor, global warming potential is only increased by 1.5-1.75% for both farm cases (R-C-NG and R-C-CHP). The exclusion of all air emissions also results in negligible change, reducing global warming potential fine particulate matter formation, and terrestrial acidification by no more than 0.6%, 1.5%, and 3%, respectively. The lowest overall emissions are seen for the case of 100% closed hydroponic systems, but again, this results in changes of <4% across all three impact categories for both conventional farm cases. The assumption of 100% open hydroponic systems results in the same percent change, but as an increase. When comparing changed results to other farm category averages, it can be seen that, in real terms, all of these assumption result in negligible change and do not influence any trends or conclusions.

Figure 65 displays results for freshwater eutrophication (D), marine eutrophication (E), water consumption (F), and land use (G). The prior two categories are influenced by the inclusion of NO_3^- leaching emissions for closed hydroponic systems, as well as the assumption of 100% open or closed hydroponic systems. It can be seen that these impact categories are influenced more by these assumptions than the prior three categories. In particular, the inclusion of leaching for closed systems results in an approximate 8% increase in freshwater eutrophication and 27% increase in marine eutrophication impacts for R-C-NG and R-C-CHP.

Larger differences in freshwater and marine eutrophication impacts are seen for the cases assuming 100% open and 100% closed hydroponic systems. This results in an approximate 17% change in freshwater eutrophication impacts and 54% change in marine eutrophication impacts for R-C-NG and R-C-CHP, with the assumption of all open systems resulting in an increase and the assumption of closed systems resulting in a decrease of these amounts. For marine eutrophication, the assumption of all open systems pushes impact levels for the conventional cases to a similar level of that of the urban farm category, which has the highest impacts. On the other hand, the assumption of all closed systems decreases marine eutrophication impacts for the conventional farm cases so that these are now lower than the peri-urban average, thus being the second lowest out of all farm categories.

Water consumption and land use are only influenced by the assumption of 100% open or 100% closed hydroponic systems, but the changes seen in these categories are largely negligible. Water consumption is influenced more, seeing an approximate 10% change based on the assumption of open or closed systems. This is because closed systems are assumed to require 30% less water than open systems, due to the recirculation of water (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018). In real terms, this difference is small and does not change overall results. Changes for land use are particularly low, with differences of only approximately 0.6%. This change in land use is a result of the lower amount of fertilisers required for closed systems, and thus has to do with land use for inputs rather than actual land use for cultivation.

Overall, the largest changes in impact levels were seen based on the applied proportions of open and closed hydroponic systems, with this particularly impacting the freshwater and marine eutrophication categories. Uncertainties around the N_2O emission factor and the inclusion or exclusion of N-related air emissions for soilless cultivation were not found to significantly affect impact levels or final results. The inclusion of nitrate leaching within closed systems did influence eutrophication impacts, but did not largely change overall results.

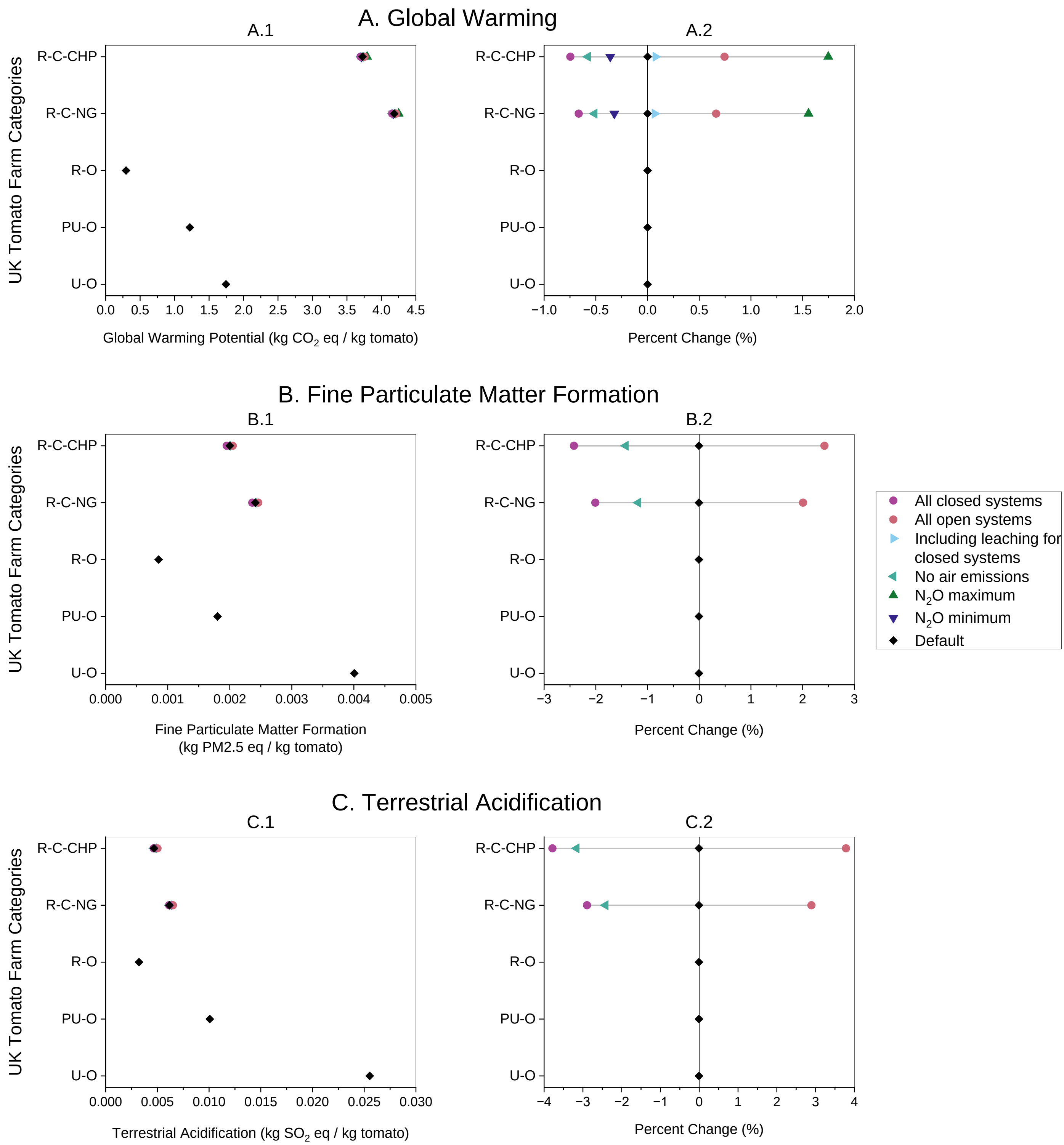

Figure 64 *- Global warming (A), fine particulate matter formation (B), and terrestrial acidification (C) impacts from sensitivity analyses applied to soilless cultivation for UK tomato production. Plots display default impact levels from Scenario 1 and the impact levels generated by each sensitivity case, which include: minimum and maximum values for the direct N_2O emission factor; the exclusion of all N-related emissions to air from cultivation (N_2O, NO_2, and NH_3); the inclusion of nitrate leaching emissions for closed hydroponic systems; and the assumption of all hydroponic systems being open or all being closed (with the default assuming 50/50). Actual impact levels from each case are displayed as farm category averages (A.1, B.1, C.1), per kg tomato. Plots depicting the percent change in average impact levels from the default for each sensitivity case are presented to next to those with actual values (A.2, B.2, C.2). The default value is depicted at 0 with a vertical*

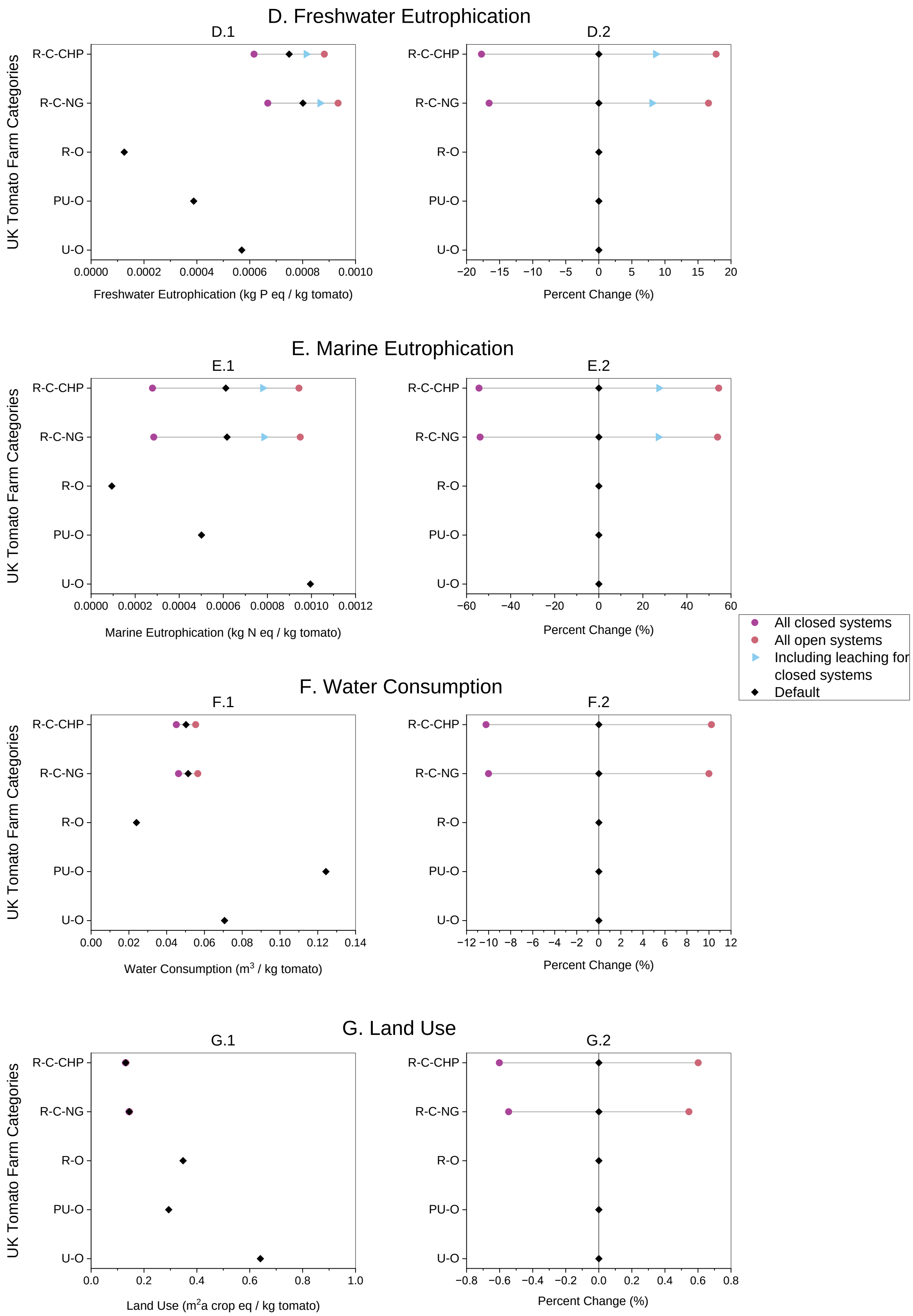

Figure 65 *– Freshwater eutrophication (D), marine eutrophication (E), water consumption (F), and land use (G) impacts from sensitivity analyses applied to soilless cultivation for UK tomato production. Plots display default impact levels from Scenario 1 and the impact levels generated by each sensitivity case, which are: the inclusion of nitrate leaching emissions for closed hydroponic systems and the assumption of all hydroponic systems being open or all being closed (with the default assuming 50/50). Actual impact levels from each case are displayed as farm category averages (D.1, E.1, F.1, G.1), per kg tomato. Plots depicting the percent change in average impact levels from the default for each sensitivity case are presented to next to those with actual values (D.2, E.2, F.2, G.2). The default value is depicted at 0 with a vertical line; a percent increase is designated as a (+)% value and a percent decrease is presented as a (-)% value.*

3.2 Discussion

Overall, this study aims to understand how farm scale, distance to consumer, and management practices influence environmental impacts for different local agriculture models, using a lifecycle assessment (LCA) methodology. Results have been presented for two crop lifecycles (kale and tomatoes) across two case study locations in different countries (Georgia, USA and England, UK), using three allocation scenarios to attribute burdens from composting and combined heat and power (see: Section 2.1.5). Impacts were assessed for germination, cultivation, processing / storage, and distribution stages in the crop lifecycles, with system boundaries including burdens from seedling production up until the point when the crop reached its final point of sale. Farms were grouped into different categories based on rurality and management practices, which included: urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C).

This research provides the most comprehensive case study analysis of environmental impacts between different scales of local agriculture to date (within the author's knowledge), including a total of 40 crop lifecycles across 11 farms in the U.S. and 14 farms in the UK. All farms included are commercial, for-profit enterprises. Attempts were made to include more than one farm in each category for each crop and country lifecycle, but this was not always possible due to the availability of farms to participate in the study. Data was taken from the same year (2019) across all farms to ensure consistency, and only farms that designated this as an 'average' or 'typical' year were considered. Conventional farms were selected to represent the industry standard where possible, including some of the top producers in Georgia and England for both kale and tomatoes. For UK conventional tomato production, which occurs in heated glasshouses, two cases of conventional production were considered: one using conventional heating (natural gas) and electricity sources (UK national grid), designated as R-C-NG, and one using combined heat and power, designated as R-C-CHP. For organic farms, the notion of a 'representative' farm is less clear as management practices vary widely depending on scale.

Throughout these LCAs, this study also aims to uncover which models of local agriculture are more sustainable from an environmental standpoint, and if this varies based on country context. Testing the notion that organic agriculture, hyper-local food production, and/or urban agriculture can improve the sustainability of food supply chains is of particular interest, due to the recent support of these strategies in the media (Garcia, 2019; Henley, 2020), from city and national governments (Greater London Authority, 2018; Warshawsky and Vos, 2019; City of Atlanta, 2022; USDA Press, 2022), and in academic literature (Pearson, Pearson and Pearson, 2010; Specht *et al.*, 2014; Reganold and Wachter, 2016; Altieri and Nicholls, 2018; Langemeyer *et al.*, 2021). However, the results from this LCA do not provide an all-encompassing answer to what the most sustainable local agricultural model might be. Instead, what was found is that the ideal model for local agriculture, in terms of lowest environmental impacts, varies based on the crop in question and the country context, as well as the specific assumptions and allocation practices used by the LCA practitioner. High variation was observed between individual organic farms, implying that differences in impacts between these farms are driven more by individual management practices rather than scale or rurality, although farm location and scale influence the management practices that can be used. The high variation in impacts between organic farms often resulted in the average impacts for certain farm categories being driven by outliers, which also makes it difficult to compare farm categories as a whole and select an 'ideal.'

3.2.1 Comparing farm categories: Scenario 1

Despite this, results did show some basic trends between farm categories for each country. For U.S. production in Scenario 1, the rural conventional farm category had the lowest average impacts per kg crop output for the majority of impact categories, seen for both tomato and kale production; these farms also had some of the highest yields. This was seen for all seven of the reported environmental impacts except marine eutrophication and water consumption, where conventional impacts were similar or slightly higher than the next lowest impacting farm category. In the UK, the rural organic farm category had the lowest average impacts per kg crop across most categories for both tomato and kale production. Exceptions were seen for land use, where the conventional farm category had lower impacts for both crop cases, and also for terrestrial acidification and water consumption in the case of kale production. The UK rural organic farms (n=2) either had comparable or higher yields than conventional farms for kale production. However, for tomato production, the conventional farm category had the highest yields by far, at 33 kg sellable tomato (gross m)$^{-2}$, due to year-round hydroponic production. Compared to other organic farm categories, the rural organic farms still had the highest yield on average (4.55 kg m^{-2}).

However, even though the U.S. rural conventional and UK rural organic farm categories have the lowest impacts for tomato and kale production on average, on an individual farm level the differences between farm categories are not as clear-cut. This is due to the high variation in impact levels observed between organic farms, both within and between farm categories. When comparing individual farms rather than category averages, in every country and crop lifecycle the lowest impacting farm across the majority of impact categories is an organic farm. Thus, while in the U.S. the rural conventional category had on average the lowest impacts, these conventional farms were individually not the overall lowest impacting farms for most impact categories.

In the U.S., the overall lowest impacting farms across the majority of impact categories are peri-urban organic farms, although not necessarily the same farm for each impact category or crop lifecycle. On the other hand, this farm category also includes some of the highest impacting farms for U.S. tomato and kale production. For UK production, the rural organic farms generally have the lowest impacts (as does the category average), but in many cases this is rivalled by other peri-urban or conventional farms. The highest impacting farms across both countries and crop lifecycles are generally urban or peri-urban farms, except for UK tomato production, where for some categories conventional production is the highest (e.g., global warming). However, these trends are also influenced by allocation methods, particularly for composting burdens, which affect certain impact categories such as global warming, terrestrial acidification, and marine eutrophication.

3.2.1.1 Literature comparisons

Global warming, acidification, and land use impacts for tomatoes and kale, as farm category averages per kg crop, were compared with global aggregated values for tomatoes and brassicas (respectively) within a recent global meta-analysis of by Poore and Nemecek (2018a). The meta-analysis included a total of 570 agricultural LCA studies, with system boundaries from production impacts up to final transport, similar to this LCA. Generally, impacts values for U.S. and UK tomato and kale production are in a similar range and order of magnitude as the 10^{th}-90^{th} percentile provided in the meta-analysis. However, for U.S. kale production, global warming and acidification impacts are generally on the higher end of those reported by Poore and Nemecek (2018a), especially for urban and peri-urban farms, although within the same order of magnitude. Similarly, land use impacts for UK kale production (1.2-4.4 m^2a per kg kale) are on the higher end of the reported 10-90^{th} percentile range (0.2-0.8 m^2a per kg brassicas). Still, this study generally produced impact values in line with previous literature, although impacts for kale were usually on the higher end of those reported for brassicas worldwide.

Only for UK tomato production has there been a prior LCA study comparing organic and conventional production for the same crop and geographical area (Williams, Audsley and Sandars, 2006), which was funded by the UK Government Department of Environment, Agriculture and Rural Affairs (Defra). This study included only on-farm burdens (i.e., excluding packaging, storage, and distribution) and used data from a wide range of sources including from national surveys, agricultural organisations, and expert advice. The conclusions from Williams, Audsley and Sandars (2006) directly contrast those in this study; they found that global warming, eutrophication, acidification, and water consumption impacts of organic production were roughly double that of conventional production when comparing the current UK market mix of both cases. This assumed a 25% inclusion of CHP in conventional production. In this study, global warming, eutrophication, acidification, and water consumption impacts for the rural organic farm category are at least half of the conventional cases. In particular, the global warming potentials of rural conventional production via CHP and conventional energy sources are respectively 13 and 14x higher than the rural organic average.

The higher impacts seen for organic tomato production by Williams, Audsley and Sandars (2006) was attributed mainly to lower organic yields, which were approximately 75% of equivalent non-organic tomato yields; this was also related to the higher use of low-yielding varieties grown in organic production (e.g., heirloom tomatoes). For this study, the average yield of the rural organic farm category (4.55 kg m^{-2}) was approximately 14% of the conventional yield (33 kg m^{-2}), making it even more difficult to determine why burdens of organic production were so much higher in the prior LCA study. The main differences can likely be attributed to the fact that Williams, Audsley and Sandars (2006) used national data to represent the market mix and industry standard; in this case, organic, field-based tomato production generally occurs as a small part of a larger, conventional (soilless) tomato operation. This is in contrast to this study, where small-scale organic farms were considered, which perhaps were able to achieve lower impacts due to low resource inputs whilst relying on highly labour-intensive production. However, in both studies, land impacts of conventional farming systems were lower, on average, than all organic systems.

3.2.2 Influence of management practices

The majority of impacts for crop lifecycles comes from the crop cultivation phase on the farm, consistent with a wide range of other LCAs on food production (Roy *et al.*, 2009; Perrin, Basset-Mens and Gabrielle, 2014; Notarnicola *et al.*, 2017; Poore and Nemecek, 2018a; Dorr *et al.*, 2021). Thus, differences between farms are mainly driven by the specific production practices on the farm, rather than 'off-field' impacts (e.g., seedling production, packaging, processing, crop storage, and distribution). Of course, the system boundaries of this study end when the crop reaches its final point of sale, and thus burdens from retail environments are not included. This presents a degree of uncertainty for conventional farms in particular, as the crops produced by these farms are sold through the supermarkets, which would have additional infrastructure, energy, and waste burdens. However, other studies evaluating environmental impacts of food items have found that contributions from the wholesale/retail environment are relatively small in comparison to on-farm processes or other food processing burdens (Muñoz, Milà I Canals and Fernández-Alba, 2010; Notarnicola *et al.*, 2017). Thus, the high contribution of the cultivation stage to kale and tomato lifecycles within the system boundary indicates the importance of management practices in driving impact differences between local farms.

3.2.2.1 Organic vs. conventional

The overarching management practices compared within this study are conventional and organic production. A wide variation in impacts was observed between organic farms, which showed some of the highest and also some of the lowest impacts within the same crop and country lifecycle. This suggests that simple designations of 'organic' or 'conventional' do not necessarily capture the sustainability of a farming system or crop output, as this depends on specific local contexts, the crops being evaluated, and specific management practices (Meier *et al.*, 2015). Certain models and management practices employed by organic agriculture *can* result in lower environmental impacts per crop produced in comparison to conventional agriculture, which has not been seen consistently among other LCAs (Tuomisto *et al.*, 2012; Meier *et al.*, 2015), but this is not always the case.

The highest variation in impact levels was generally seen within the peri-urban farm category, which contained some of the highest and some of the lowest impacting farms across impact categories, for all country and crop lifecycles. In terms of production practices, peri-urban farms can be seen to lie 'somewhere in the middle' of the small-scale, intensive nature of urban agriculture and the larger field-scale nature of rural agriculture. Thus, some peri-urban farms were of a larger scale and adopted practices more similar to rural organic farms, whilst others were characterised by small-scale household operations and thus adopted practices more similar to urban farms (e.g., intensive market gardens). The large range of variation can particularly be seen for U.S. kale production, U.S. tomato production, and UK kale production, where global warming impacts for the peri-urban farm category respectively ranged from 0.48-12.0, 0.58-2.5, and 0.4-3.8 kg CO_2 eq (kg kale)$^{-1}$. However, it should be noted that high variation might have also been observed for the urban farm category (and indeed was seen for the two urban farms considered for UK tomato production), if more sample farms were included within this category.

In comparison, relatively low variation in impact levels was observed between conventional farms. However, there was also a smaller number of conventional farms assessed for each crop and country lifecycle in comparison to the total number of organic farms assessed. For U.S. kale production in particular, only one conventional farm was evaluated due to the difficulty of recruiting farmers in this area. For UK tomato production, again only one farm was used as a case study, although certain data was based on industry averages (e.g., yield) or literature sources, due to the data-protective nature of the controlled environment agriculture industry. However, two conventional farms were included for U.S. tomato production and five for UK kale production; for these cases, similarities were generally observed in impact levels between conventional farms. This is due to the relatively standardised and similar production practices used on these farms, especially for seen for tomato production in Georgia, as production practices are quite standardised throughout the southeast U.S. region (UGA Extension, 2017).

The high variation of impact levels between organic farms in this study provides further evidence for why the 'organic vs. conventional' debate has been so highly contested and resulted in such different outcomes among LCA studies (Tuomisto *et al.*, 2012). A wide range of studies on vegetable cropping systems have found that conventional agriculture generally results in lower impacts per crop output than organic agriculture (Venkat, 2012; Bos *et al.*, 2014; Foteinis and Chatzisymeon, 2016), although there are studies that have also concluded the opposite (De Backer *et al.*, 2009; Lindenthal *et al.*, 2010; Zafiriou *et al.*, 2012). In cases where conventional production had generally lower impacts than organic, this was mainly attributed to the higher yields; this is why organic production tends to have lower impacts when assessed with area as the functional unit, whilst results are more variable when assessed based on crop output (Nemecek *et al.*, 2011; Meier *et al.*, 2015). However, in this study, there were many organic farms that achieved yields similar to or even higher than conventional farms, for all cases except UK tomato production, where hydroponic production results in extremely high yields. In addition, lower impacts were not always seen on the farms with higher yields; indeed, in some cases farms that had both low resource inputs and low yields still achieved some of the lowest impacts (e.g., PU-O-3, U.S. kale production). This concurs with Dorr *et al.* (2021)'s review and meta-analysis of LCAs on urban and peri-urban agriculture, which found no strong correlation between yield and climate change impacts.

Obviously, the environmental viability of organic agriculture depends on the specific crop, region, and impact category in question (Nemecek *et al.*, 2011; Meier *et al.*, 2015), but, as seen in this study, also on the specific farm cases selected for examination. Many past studies have used averaged data for a specific region or country that may have been provided through universities, research centres, industry data, or national datasets (Williams, Audsley and Sandars, 2006; De Backer *et al.*, 2009; Lindenthal *et al.*, 2010; Venkat, 2012), while others defined specific model farm cases (Bos *et al.*, 2014) or used actual information provided by individual farm case studies (Zafiriou *et al.*, 2012; Foteinis and Chatzisymeon, 2016), like in this LCA. When using averaged or industry data, information about the variation within these systems is lost (Nemecek *et al.*, 2011); on the other hand, when using case studies, the farms selected might sway results and not be representative of the farming system as a whole (Bos *et al.*, 2014). Thus, care should be taken when generalising results using data of either type, due to the great variability often observed between individual farms, even within 'organic' and 'conventional' designations (Guerci *et al.*, 2013). For this study, it is particularly important to highlight that it is almost impossible to select a 'representative' local organic farm due to the wide variation of organic farming models that exist, even within a relatively small geographical area (e.g., one U.S. state). This is even harder to define for an urban or peri-urban farm, due to the relatively small number of these farms that exist in the first place. However, the benefit of analysing a wide array of case study farms is that the specific management decisions that result in higher or lower impacts can be identified, bringing the opportunity to understand what practices are driving differences between farms rather than assuming differences based on simple ‘organic’, ‘conventional’, ‘urban’, or ‘rural’ classifications.

3.2.2.1.1 Main impact drivers for organic farms: the importance of compost

Organic farms in this study used a wide range of production practices and inputs, both within and between countries. Major differences in the lifecycle inventories between organic farms were observed for: the types of organic fertilisers used (e.g., compost, manure, or processed products); the employment and length of crop rotations; the use of cover crops; the cultivation of crops in polytunnels or in fields; cultivation using intensive, market garden production or field-scale production; the use of tractors and other machinery; and the use of irrigation. Although many different practices varied between individual organic farms, the main process driving impacts, and thus the variation in impacts, was compost production and use.

Compost was used as an input by the majority of organic farms in this study, especially for tomato production. For impact categories such as global warming, fine particulate matter formation, and terrestrial acidification, compost-related impacts came mainly from the burdens of producing compost – particularly, the N_2O, CH_4, and NH_3 emissions generated from decomposing organic matter in the compost pile (see: Table 43 and Table 44 for emission inventories). For marine eutrophication, impacts were driven mainly by compost application emissions, due to estimated nitrate leaching. The differences in impact levels between organic farms for these impact categories are driven by the varied amounts of compost applied, which ranged anywhere from 0-29 kg compost per cropped m^2 among all organic farms.

Higher levels of compost application were generally observed on the smaller-scale urban and peri-urban farms, and thus these farms generally had higher impacts. This concurs with other studies that have also found relatively high rates of compost application on urban farms and gardens, which has been identified as a potential source of leaching (Small *et al.*, 2019; Wielemaker *et al.*, 2019; Shrestha, Small and Kay, 2020). The reasons for higher compost application rates on these farms are unknown, but can be hypothesised. Smaller, urbanised farms usually grow produce on a market garden scale (e.g., in raised beds, small plots or polytunnels), where compost is often used to fill or make beds (Altieri *et al.*, 1999; Goldstein, 2009; Ackerman, 2012). Compost application has been identified as an important practice especially for more urban soils, where soil quality or contamination are often issues (Beniston and Lal, 2012). In these cases, compost can be used to improve soil fertility and quality and can also provide a direct barrier from contaminated soils, or help remediate them by stabilising certain heavy metals (Lepp, 1981; Brown and Jameton, 2000; Wortman and Lovell, 2013). Further, compost can help improve soil structure and reduce compaction (Logsdon, Sauer and Shipitalo, 2017), which was cited as an issue by some urban and peri-urban farmers in this study. Because of these benefits, compost may be applied in high amounts on urban and peri-urban farms, especially when this resource is more available due to either composting on-site or from access to municipal compost from city waste. Indeed, some farms in this study used composting as the waste management solution for other farm processes (e.g., chicken manure) and for on-site households; thus, large amounts of compost were readily available and easily accessible.

In contrast, larger organic farms, which are usually found in rural areas, can grow crops on a field scale and may use cover crops and other organic fertilisers that are more commonly found in agricultural areas (e.g., manure and by-products from other crop production). Large amounts of compost are generally not applied on this field scale, especially if not produced on the farm, as this can be an expensive input. This was seen especially in the case of kale production in the UK, where farms growing on a field scale did not apply any compost and relied on cover crops and manure for fertility instead. As a result, these farms had lower impacts compared to most other organic farms that grew kale on a smaller scale and used compost as the main input. However, smaller scale farms that applied moderate levels of compost and were able to offset this with higher yields also had relatively low impacts.

Despite the importance of compost as the main determining factor driving impacts and impact variation between organic farms in this study, other LCA studies comparing organic and conventional agriculture have generally not identified this as the case. The reasons for this vary. Compost may not be a typical input for organic farms in other countries or for other crops; indeed, this was not considered as an input in Bos *et al.* (2014)'s analysis of organic vegetable farms in the Netherlands or De Backer *et al.* (2009)'s study on organic leek production in Belgium. Additionally, Zafiriou *et al.* (2012) identified that the high variation in primary energy and greenhouse gas impacts observed between organic farms in Greece was mostly related to the differences in fertilisers used, but compost was not specified as the input driving this. On the other hand, there is evidence that compost production and use is an important practice on organic farms in the U.S., employed by 35% of organic farms in the country (USDA, 2019a). This is generally a more common practice on smaller-scale organic farms (<40 ha) (Liebert *et al.*, 2022), such as those in this study. Indeed, in this study the largest area of cultivated organic land is 8 acres (3.2 ha) for farms in Georgia and 27 acres (10.9 ha) for farms in England; this is in contrast to the average organic land area per farm being 70 acres (28.3 ha) in Georgia and 334 acres (135 ha) across the U.S. as a whole (USDA, 2019a). Thus, other LCA studies may have relied on larger-scale farm case studies or aggregated country-wide farm survey data, which lends itself toward larger-scale production, where compost use is not as common.

In contrast, compost production has been cited as a major impact contributor for peri-urban farms in a recent review and meta-analysis (Dorr *et al.*, 2021). However, this review also highlights that modelling compost production burdens is a major area of uncertainty throughout these LCAs. There is a lack of detail regarding how and if compost production burdens are modelled throughout LCAs that regard compost as a farm input, especially in terms of the fugitive emissions generated from the compost pile. For example, in Venkat (2012)'s comparison of 12 organic and conventional crops in California, compost production and transport was identified as a major contributor to greenhouse gas emissions for crops that used high levels of compost inputs, particularly walnuts and almonds. This study specified inclusion of compost production within the LCI, particularly mentioning water and energy inputs. However, it is not clear if composting emissions (decomposition emissions) were explicitly included in this model. The study reported that the total greenhouse gas emissions associated with compost production, including energy burdens, as 0.07 kg CO_2 eq (kg compost)$^{-1}$; for this study, just the fugitive CH_4 and N_2O emissions associated with industrial composting in windrows equate to 0.157 kg CO_2 eq (kg compost)$^{-1}$, when applying ReCiPe characterization factors of 34 and 298 kg CO_2 eq (kg gas)$^{-1}$, respectively. Thus, it can be assumed that fugitive emissions were likely not accounted for in the study by Venkat (2012), and this is possibly why compost was not identified as a major impact driver for other crops (e.g., the vegetables examined). However, this may also have been due to the less common use of compost on larger-scale organic farms, which predominate in California (average size 320 acres / 130 ha) (USDA, 2019a).

In other cases, compost production burdens may not be explicitly modelled, even if compost is used on the farm, perhaps because it is not regarded as a sensitive factor within the LCA (despite this study identifying otherwise). For example, compost production was not identified as a major contributor to global warming potential impacts for fruit and vegetables produced on a community farm near London (Kulak, Graves and Chatterton, 2013). However, this study mainly relied on data produced from national averages and assessments, or on global warming figures from crop cultivation in general, as provided by other LCAs. Thus, there was no detailed discussion of the specific processes contributing to impacts within the crop cultivation phase, and compost production was not specifically modelled, despite its use on the case study farm.

In terms of compost application burdens, Dorr *et al.* (2021)'s review of urban agriculture LCAs highlights that the inclusion of direct agricultural emissions from organic fertilisers is often inconsistent or not transparent, which again may divert attention from compost (or organic fertiliser application in general) as an important impact driver for organic farms. This highlights that perhaps compost is not normally treated as an area of importance or sensitivity in LCAs on organic or urban agriculture, but this study shows that it should be.

Additionally, the application of varied allocation procedures to the composting process means that LCAs can draw varied conclusions about compost impacts (Dorr *et al.*, 2021). When production burdens for compost inputs were not allocated in this study, seen within allocation Scenario 2, many of the marked differences between organic farm impact levels dissipated, especially for terrestrial acidification, fine particulate matter formation, global warming potential, and to a lesser extent, freshwater eutrophication impacts. This implies that studies that do not include compost production burdens may not uncover this variance between organic farms.

Several studies also subtract avoided burdens from composting (Dorr *et al.*, 2021), either from the avoided use of mineral fertilisers or the avoided municipal waste process (Martínez-Blanco *et al.*, 2009; Cleveland *et al.*, 2017), the latter of which was explored within this study in allocation Scenario 3. However, the other studies applying these avoided burdens did not provide context of what the impacts would be otherwise (Martínez-Blanco *et al.*, 2009; Cleveland *et al.*, 2017), making it difficult to draw out the importance of compost as a key sensitivity for organic farms. Indeed, when the avoided burdens of the municipal waste process were applied within this study in Scenario 3, the trends observed between farm category impact levels were often reversed, especially for the global warming and marine eutrophication impact categories. In these cases, the farms applying the highest levels of compost now saw some of the lowest impacts, even negative in some cases, thus essentially erasing the impact contributions of compost. In addition, several LCA studies also include the C sequestration from compost application (U.S. EPA, 2006; Recycled Organics Unit, 2007; Blengini, 2008; White, 2012; Saer *et al.*, 2013), which was explored as a sensitivity in this study. When the maximum values of C sequestration for compost were applied (based on values by Saer *et al.* (2013)), this also resulted in negative global warming impacts for some farms. This shows that the benefits of compost application may outweigh production and application burdens for certain impact categories, but that it is important to distinctly quantify what these benefits are and how they should be applied in an LCA.

Although compost production and application is indeed a major driver of impacts on organic farms, this was not the sole driver. Other important impact drivers for organic farms were: production of other organic fertilisers and associated application emissions, especially for U.S. organic farms, which used a wider array of fertilisers and less manure than UK farms; polytunnel infrastructure; cultivation machinery and operations, mainly from fuel combustion, for farms using tractors; and energy uses during cultivation and packhouse operations, especially for cold storage of crops. Fertiliser production, direct agricultural emissions, fuel use, and polytunnel infrastructure (when used) have also have also been identified as impact ‘hotspots’ for organic vegetable cultivation (De Backer *et al.*, 2009; Venkat, 2012; Bos *et al.*, 2014; Adewale *et al.*, 2016) or urban and peri-urban agriculture (Kulak, Graves and Chatterton, 2013; Rothwell *et al.*, 2016; Dorr *et al.*, 2021). Additionally, transport was found to be a major contributor to impact for some peri-urban and rural organic farms within the U.S., which is discussed in greater detail in Section 3.2.3.1. Yield was not always a major determinant of impacts for most categories, but was seen to be important for marine eutrophication impacts and land use impacts. For the prior, this is because eutrophication is mainly driven by leached nutrients; thus, even if P and N application rates are low or moderate per area, impacts will be high per crop output if a farm has relatively low yields. For the latter, this is because land use impacts are driven mainly by direct land uses for cultivation, and thus are directly tied to yields when assessed per crop output.

3.2.2.1.2 Main impact drivers for conventional farms

For field-based conventional production (i.e., U.S. kale, U.S. tomato, and UK kale), the main impact drivers were generally similar, with some minor differences. For global warming, fine particulate matter formation, and terrestrial acidification, impacts came mainly from fertiliser production (due to the high energy requirement for manufacturing synthetic N fertilisers) and the agricultural emissions associated with fertiliser application. This is consistent with Poore and Nemecek (2018a)’s meta-analysis of agricultural LCAs, which cited the major contributors to on-farm global warming and acidification impacts for vegetable production as direct agricultural emissions from fertiliser application, fertiliser production, and electricity and fuel use. In this study, however, cultivation machinery and operations (including fuel use) are generally not a significant contributor, with the exception of certain UK kale farms. Energy use from cultivation is also not a major contributor, although energy use in packhouse operations (mainly from cold storage) is a driver of impacts for U.S. and UK kale production, specifically for freshwater eutrophication impacts. This is mainly driven by emissions associated with coal power within the electricity mix.

Marine eutrophication impacts are driven almost exclusively by agricultural emissions, mainly from nitrate leaching from fertiliser application, which has also been seen in other LCAs on vegetable crop lifecycles (De Backer *et al.*, 2009; Poore and Nemecek, 2018a). Levels of total N applied per cropped area was similar between organic and conventional farms, as also found by Bos *et al.* (2014) for intensive organic and conventional farms in the Netherlands. This is why marine eutrophication impacts were not necessarily higher for conventional vs. organic farms in this study, and impacts for conventional farms tended to be lower than the organic farms applying the highest levels of compost. However, this is also because the same nitrate leaching model was applied to total N from all sources (i.e., mineral and organic fertilisers), which presents a degree of uncertainty; when a sensitivity analysis was applied considering leaching only from soluble N, average marine eutrophication impacts for all organic farm categories became lower than or similar to respective conventional averages.

For UK conventional tomato production, impact drivers were different due to the use of hydroponic cultivation in heated glasshouses. Energy use from heating glasshouses constituted the majority of global warming impacts, for both the case of heating with natural gas (R-C-NG) and with CHP (R-C-CHP). This is consistent with a wide range of other studies on hydroponic tomato production in heated glasshouses, which identified energy use from heating as the main contributor (≥80%) to both on- and off-farm impacts of global warming (Williams, Audsley and Sandars, 2006; Antón *et al.*, 2012; Torrellas, Antón, Ruijs, *et al.*, 2012; Röös and Karlsson, 2013), when considering natural gas heating or CHP using energy allocation. However, these studies also identified heating to be the main contributor to on-farm acidification and eutrophication impacts (≥80%), which was not seen as much in this study, as similarly high contributions came from farm infrastructure; however, this could be because of the inclusion of packhouse infrastructure, which was not included in most other studies.

The use of CHP was found to reduce global warming impacts by 11% when considering allocation by energy (Scenario 1 and 2) and by 24% when applying the avoided burdens from the production of surplus electricity through the national grid (Scenario 3). This is similar to the study by Williams, Audsley and Sandars (2006), which found that employing CHP across all UK conventional tomato production would result in a 17% decrease in global warming potential, when applying avoided burdens. However, Williams, Audsley and Sandars (2006) also found that this resulted in overall negative acidification and eutrophication impacts, which in this study was only seen for the prior. This is likely due to both the fact that this study had a wider system boundary, whereas Williams, Audsley and Sandars (2006) considered only on-farm processes. Also, the national electricity grid make-up in Great Britain has changed since the prior study was completed. For example, coal represented 17% of the UK inland energy consumption in 2004 and only 3% in 2019 (UK Department of Trade & Industry, 2005; UK Department for Business, Trade & Industrial Strategy, 2020).

3.2.3 Influence of scale and location

Although individual management practices largely determined impacts on farms, in some cases farm scale, degree of locality (i.e., urban, peri-urban, and rural), and geographic location can determine the set of management practices available to a farmer. Thus, while impacts differences between farms cannot be boiled down to a specific urban, peri-urban, or rural context, this can influence what practices drive impacts on farms.

3.2.3.1 Economies of proximity: transport impacts

One of the main benefits of urban agriculture is cited as reduced food miles (Mok *et al.*, 2014; Specht *et al.*, 2014), and thus theoretically lower burdens associated with transport compared to the conventional food supply chain (Kulak, Graves and Chatterton, 2013). In this study, burdens from transport were found to be negligible contributors to environmental impacts for all urban farms, as well as for many peri-urban farms, especially when all produce was sold on-site or through nearby markets and restaurants. However, other impacts from cultivation (mainly compost production and application) meant that impacts from urban farms were usually some of the highest compared to other farm categories. This implies that while urban farms indeed have reduced impacts from transport in comparison to other locally-produced conventional foods, this does not necessarily mean overall lower impacts if cultivation processes require high resource inputs.

Transport burdens played a more significant role (in real terms) to impacts for other non-urban farms, particularly in the U.S. rather than the UK, due to larger transport distances for the prior. Contributions in percent terms were similar between U.S. rural organic and rural conventional farms, despite the much longer distances that food travelled from conventional farms, as it was often sold in other states in the southeast or even along the entire East coast. However, in real terms, transport was a minor contributor to impacts for U.S. conventional farms, because overall impacts for these farms were generally some of the lowest. In addition, even though food travelled the farthest from U.S. conventional farms compared to all other farms in this study, this food travelled efficiently, in fully packed, fuel-efficient lorries. On the other hand, transport impacts were relatively high for rural organic and some peri-urban farms because of the inefficient modes of transport used, seen as food travelled relatively long distances (50-200 km) in vans that were not necessarily fully-packed. This was not seen for UK rural organic farms due to the more urbanised make-up of England vs. Georgia; UK rural organic farms sold produce into the closest towns or cities, which were numerous, while the rural organic farms in Georgia had to sell both in local towns and into the main urban centre of Atlanta to reach an appropriate number of consumers.

This implies that a reduction in food miles does not always translate to a reduction in transport impacts when the methods of transport are not efficient. Additionally, farther transport distances for larger-scale organic farms can mean that crops must be harvested days before transport, potentially resulting in further burdens from crop storage. Considering this, it can be theorised that 'economies of proximity' exist for small-scale, local agriculture; that is, there is an ideal distance between small-scale farms and points of sale that can result in relatively low post-farm impacts, despite the more inefficient modes of transport that may be used. For U.S. rural organic farms, it is clear that the distances travelled to sell into Atlanta were not environmentally viable, although these sales were likely important for the farms financially.

Other studies have also identified that transport is not typically a main impact contributor for large-scale, conventional production, like in this study, or even for many types of imported foods (Weber and Matthews, 2008; Williams *et al.*, 2009; Poore and Nemecek, 2018a). However, high contributions from transport have also been seen in other studies when evaluating localised production within one country. For example, transport burdens were identified as important contributors of global warming impacts by Meisterling, Samaras and Schweizer (2009), when comparing production and transport of organic and conventional wheat within the U.S. Further, Rothwell *et al.* (2016) found that packaging and transport to market constituted the majority of global warming impacts for an interstate farm transporting produce 930 km to market in Sydney, Australia; although the global warming impact from crop cultivation for this farm was lower than the peri-urban farms evaluated in the study, when taking post-farm processes into account, this caused this farm to have some of the highest global warming impacts. It is clear that in this case, economies of proximity were not achieved.

3.2.3.2 Economies of scale

Economies of proximity can be outweighed by economies of scale, or the advantages that are achieved in terms of efficiency when producing goods at a larger scale (Silberston, 1972). Considering environmental impacts, this means that as a farm grows in size, the levels of production that are achieved can outweigh environmental costs associated with both on-farm burdens, such as fertiliser use, soil cultivation processes, and energy use, as well as off-farm burdens, such as transport. Larger scale also generally means that farms can invest in more fuel and energy-efficient equipment, such as tractors and irrigation systems, and generally that equipment is more fully utilised (e.g., across a range of crop processes) (Faris, 1961). Indeed, a commonly cited disadvantage of urban agriculture, or small-scale agriculture more generally, is the lack of appropriate equipment for small-scale production (Kennard and Bamford, 2020).

Economies of scale also explain why both U.S. and UK conventional farms did not have high impact contributions from transport in real terms. Although these farms transported produce farther distances than other farm categories, this transport was efficient as large amounts of produce were transported at one time, and thus the environmental cost per crop output was relatively low. Additionally, in the U.S., the relatively high yields and associated economies of scale achieved by conventional farms resulted in these farms having some of the lowest impacts. This is also because the organic farms had relatively high impacts due to high resource use per crop output. On the other hand, UK rural conventional farms generally had higher impacts than UK rural organic farms because the latter were able to achieve relatively high yields with low or moderate cultivation inputs.

Even on the smaller scale of the organic farms, economies of scale played a role in some instances. This was seen from the low overall utilisation of tractors and machinery on some smaller farms, which resulted in higher impact contributions from the production burdens of these items. This is in contrast to other, larger-scale farms that used tractors over a wider area and for more crops, thus reducing the allocation of these items to singular crop lifecycles (e.g., kale). Indeed, the utilisation level of farm equipment has been identified as one of the main processes driving economies of scale for crop production (Faris, 1961).

3.2.3.3 Local context

Although differences in impacts between farm categories was related more to individual management practices than anything else, local context can play a role in impacts by determining the resources available to a farm or by imposing additional challenges that must be addressed. For example, the local geographical context played a role in the modelling of nitrate leaching models, as the model used in this LCA required inputs based on climatic and soil conditions (de Willigen, 2000). This particularly drove the differences in marine eutrophication impacts between UK kale farms, due to the more varied precipitation levels observed between regions in England (as an island country) than Georgia. Conditions that may lead to more leaching (such as higher precipitation or lower soil clay content) or runoff (such as field slope) should be thus considered when selecting fertiliser inputs to use and deciding when to apply them. Although it is likely that this was considered by farmers in this study, certain steps that may be taken to minimise leaching and runoff (e.g., by minimising soil cultivation, relying on cover crops, or only applying fertilisers at certain times of the year) are not captured within many of the nitrate leaching models commonly used in LCAs. This represents an area of uncertainty within this LCA, as well as many others, as the benefits of certain management practices are lost (Venkat, 2012).

Farms may also face other challenges based on their specific local context. As has already been discussed, urban and peri-urban farms particularly may be faced with poor quality or contaminated soils that may require regeneration or remediation (Beniston and Lal, 2012; Beniston, Lal and Mercer, 2016), and thus likely influenced the high levels of compost applied on these farms. This highlights an important trade-off for urban agriculture – the ability to produce food closer to the final consumer, but potentially at the cost of accessing high quality soils as well as high quality inputs (Kennard and Bamford, 2020).

Additionally, the location of the farm in relation to densely populated areas can be an issue. This has already been identified in terms of the additional transport burdens for U.S. rural organic farms, where their produce travels 77-183 km to Atlanta for sale. However, this also determines the types of sale methods that can be employed by these farms; while some urban and peri-urban farms were able to operate CSA or delivered ‘veg box’ models, this was not possible for rural farms in the U.S., as they did not have access to large numbers of consumers. In some cases, these farmers identified that high amounts of waste were due to the fact that not all produce could be sold at market or to restaurants, which is less of an issue within CSA or veg box models (as more produce can be given to consumers). The rural location also presented an issue for some U.S. organic farms in terms of labour availability, which in some cases was also identified as contributing to crop waste through unharvested produce. One farmer in particular highlighted how difficult it was to secure apprentices or other consistent labour in rural areas. This is in contrast to many urban and peri-urban farms, which, despite the smaller-scale, often had similar numbers of employees as the more rural farms, due to the wider labour pool available in metropolitan areas. Many urban and peri-urban farms also relied on volunteer labour, which allowed for more intensive production, in comparison to rural farms that were mostly unable to access this free resource.

3.2.3.4 Country context

Geographical context can also influence farm impacts when viewed on a wider, country scale. In this study, UK organic farms were generally seen to have lower impacts for global warming, fine particulate matter formation, acidification, and eutrophication when compared with the same farm categories in the U.S. Conventional kale production generated largely similar impacts in both countries, whilst UK conventional tomato production had generally higher impacts, due to the energy-intensive hydroponic cultivation.

It can thus be seen that climate in particular plays a role in determining what types of production practices can be used, and what types of inputs are needed. Hydroponic cultivation in heated glasshouses has predominated large-scale tomato cultivation in the UK, due to the country’s suboptimal climate for growing tomatoes, as heat- and sun-loving crops (Bauer, Barney and Robbins, 2009). This is in comparison to the warmer climate of Georgia, which allows for field production of tomatoes. In the UK, the used of controlled environment agriculture allows for year-round local production, but also results in large energy inputs.

This high energy use actually results in UK-grown tomatoes having 4x higher primary energy use and 3x higher global warming impacts in comparison to imported Spanish tomatoes grown in polytunnels, when including production, processing, storage, and transport burdens (Williams *et al.*, 2009).

On the other hand, the high precipitation rates in the UK allow for kale cultivation to take place in many regions without irrigation, a practice that is required for U.S. kale cultivation. Additionally, cold storage of tomatoes is a more common practice for conventional production in the UK compared to the southeast U.S., where tomatoes are generally harvested directly into cardboard boxes in lorries. However, organic farms in the U.S. generally used more energy for the cold storage of kale than in the UK, as the colder climate negated the need for cold storage throughout the whole harvest season for this crop.

This highlights how geographical context influences the management and production practices that can be used, and thus creates trade-offs based on areas of food production. In some cases, like the case of UK vs. imported Spanish tomatoes, more optimal growing conditions can outweigh burdens of transport for imported foods (Williams *et al.*, 2009). These trade-offs are thus important to understand when evaluating the sustainability of food supply chains within countries or cities as a whole (Blanke and Burdick, 2005; Williams *et al.*, 2009; Webb *et al.*, 2013; Theurl *et al.*, 2014; Payen, Basset-Mens and Perret, 2015).

3.2.4 Impact trade-offs

One of the main benefits of LCA is the ability to evaluate trade-offs that occur between different types of environmental impacts (Finnveden *et al.*, 2009; Henryson *et al.*, 2020), which are not captured within simple primary energy assessments or carbon accounting frameworks. It is particularly important to evaluate impacts trade-offs within a local context, where certain environmental issues may be more of a concern. For example, global warming impacts are a more immediate concern for certain areas and regions, as drylands and coastal areas (Nicholls and Lowe, 2004; Jacob *et al.*, 2018; Koutroulis, 2019). Issues of water scarcity are more concerning in areas facing drought, which recently has become a more pressing issue in England; indeed, in 2022, southwest England experienced the driest conditions in nearly 90 years (Environment Agency, 2022). Additionally, land use may also be more important for countries that have limited remaining land available to use for agriculture, such as in England, versus countries that still have expansive areas of uncultivated land, like the U.S. For example, agriculture currently comprises 69% of land use in England (Defra, 2022a) and only 27% of land use in Georgia, USA (USDA NASS, 2022), with England also having a much higher extent of urbanised area (U.S. Census Bureau, 2010b; Natural Environment Research Council, 2017; Office for National Statistics, 2019).

However, additional trade-offs can occur that are not as eloquently captured within LCAs. For example, farms may employ certain management practices (e.g., conservation methods, min-till or no-till) that result in environmental benefits not captured in existing emissions models or LCA frameworks. Certain ecosystems services that are commonly associated with organic or agro-ecological production, such as enhancing biodiversity and soil health, are also not commonly assessed within LCAs (Boone *et al.*, 2019; van der Werf, Knudsen and Cederberg, 2020). Farms may also contribute to social and economic benefits in their local communities, which are not captured within basic environmental impact assessments.

3.2.4.1 Environmental impact trade-offs in LCA

In this study, significant trade-offs between the environmental impact categories included within the LCA are observed both for farm categories as a whole, as well as for individual farms. On average, organic farm categories had higher land uses than conventional farms. This is due in part to the generally lower yields seen on organic farms in comparison to conventional farms, an issue well-cited in literature (De Ponti, Rijk and Van Ittersum, 2012; Seufert, Ramankutty and Foley, 2012), as well as the additional land uses for cover crops that are more common within organic agriculture. Williams, Audsley and Sandars (2006) also found that land use was always higher for organic production compared to conventional production in the UK, when assessing ten key food commodities including bread wheat, potatoes, oilseed rape, tomatoes, beef, pig meat, sheep meat, poultry meat, milk and eggs.

Trade-offs between land use and other impact categories are vividly portrayed within UK kale production. In general, the individual farms with the lowest impacts were the organic farms that utilised field-based production methods and used mainly cover crops and manure as fertility inputs. Cover crops in particular did not contribute significantly to most impact categories, whereas fertility inputs used on other farms, such as compost or mineral fertilisers, did. However, the use of cover crops was a significant contributor to land use impacts, contributing to nearly 50% of these impacts for farms where this was the only fertility input. Additionally, farms using cover crops as the main fertility input tended to have lower yields than organic farms applying other organic fertilisers, like compost. Thus, it can be seen that the use of cover crops represents a clear trade-off between land use and other impact categories, as well as productivity. In places or situations where maximising crop output is important, the use of additional fertilisers in conjunction with short-term cover crops or green manures may be a more ideal approach. On the other hand, the use of cover crops also results in many other environmental land benefits that are not captured within this LCA framework, such as conserving soil moisture, reducing soil erosion, enhancing carbon sequestration, improving soil aggregate stability, decreasing weed and pest pressure, and even reducing particulate matter emissions from wind and machinery use (Blanco-Canqui *et al.*, 2015; Adetunji *et al.*, 2020; Blanco-Canqui and Ruis, 2020).

UK conventional tomato production had opposite trade-offs between land use and other environmental impacts, as compared to organic systems. The high energy use needed for heating glasshouses led to relatively high global warming impacts, but also resulted in much lower land uses due to the higher yields achieved from controlled environment agriculture, in comparison to UK organic production and also conventional production in the U.S. Indeed, UK conventional production resulted in yields 11x higher than the UK organic average and 8x higher than the U.S. conventional average. Barbosa *et al.* (2015) similarly identified this trade-off between natural resource use and energy use within the hydroponic production of lettuce, where hydroponic production provided 11x higher yearly yields and 12.5x lower water use in comparison to field-based production, but also required 82x more energy. While lower water consumption is often cited as a benefit of hydroponic production (Molden, 2007), this was not seen in the study, as some peri-urban and rural organic farms actually had lower water consumptions (per crop output) than hydroponic production. However, reduced rates of water consumption for hydroponic production may be less pronounced for certain soilless cultivation techniques or for certain crops, like tomatoes; indeed, Verdoliva *et al.*

(2021) reported only a 20% maximum decrease in water use for tomatoes grown in soilless media with drip irrigation compared to soil.

The trade-off between land use, other environmental impacts or benefits, and food production is classically captured within the ‘land sharing’ vs. ‘land sparing’ argument (Lin and Fuller, 2013; Muller, Ferré, *et al.*, 2017). High-tech production methods such as controlled environment agriculture and hydroponics tend to have high resource inputs in terms of energy and materials, but achieve such high yields that this can be outweighed when assessing environmental impacts in terms of crop output. The lower land use required per crop output for these technologies, seen in this study with UK hydroponic tomato production , theoretically means that land elsewhere can be ‘spared’ and left for natural purposes. This can also be the case for other highly-intensive, conventional agricultural systems. On the other hand, organic agriculture or other agro-ecological approaches that tend to produce lower yields require larger land areas to produce the same amount of food as more intensive, conventional modes of agriculture; however, the idea is that this land is managed less intensively, with less inputs, and is ‘shared’ with nature, for example by leaving more spaces for wildflower meadows or woodlands (Muller, Ferré, *et al.*, 2017; van der Werf, Knudsen and Cederberg, 2020). Many LCAs have thus found that organic agriculture results in lower impacts when considering land area as the functional unit, because of generally lower input use per area, but higher impacts when considering crop output as the functional unit (Tuomisto *et al.*, 2012).

Because of this, there has been academic debate about whether land or crop output should be considered as a functional unit for agricultural LCAs, especially when comparing organic and conventional agriculture. Production area is considered to be more appropriate when the focus is related to environmental impacts in a specific area or region (De Backer *et al.*, 2009), with the aim to provide information for regional planning and land use policies (Berlin and Uhlin, 2004). However, others suggest that product output (e.g., crop mass) should be used as the functional unit when it is important to account for production and land use efficiency (Bos *et al.*, 2014). For this LCA, product output is considered as the functional unit because the aim was to compare different agricultural models based on the ability to produce food with lower environmental impacts, with production efficiency being an important measure to meet the food needs of a rapidly growing world population (FAO, 2009).

3.2.4.2 Pesticides and toxicity impacts

Although toxicity impact categories were not a main focus of this LCA study, they will be briefly discussed here to provide further context to additional impact trade-offs as related to ecosystem and human health. In the context of agricultural LCAs, toxicity impact categories are mainly driven by potential exposures to heavy metals or pesticides through air, water, soil, or food pathways, as well as from the potential toxic effect of the emissions released from during energy use and fuel combustion (De Backer *et al.*, 2009).

Generally, it would be assumed that conventional agriculture would always result in higher ecotoxicity impacts (i.e., terrestrial, freshwater, and marine ecotoxicity) and human-related toxicity impacts (i.e., human carcinogenic and non-carcinogenic toxicity) due to the prevalent use of non-biological pesticides, which are not allowed in organic agriculture. Indeed, De Backer *et al.* (2009) found that human and terrestrial ecotoxicity impacts were at least an order of magnitude higher on conventional leek farms than organic ones. However, this is not consistently seen throughout this LCA. In the U.S., the conventional farms generally had the lowest toxicity impacts (in kg 1,4-DCB per kg crop) when compared to other organic farm category averages for the same crop type, as was the case for most other impact categories. On the other hand, conventional UK kale production had the highest environmental ecotoxicity impacts compared to organic farm category averages, although relatively lower for human toxicity impacts. For UK tomato production, the conventional farm cases had the highest toxicity impacts for all relevant categories except human non-carcinogenic toxicity, even though this farm used only biological pesticides. However, it should be noted that, in Scenario 3, all conventional farms tended to have relatively high toxicity impacts in comparison to organic farms, showing the benefit of composting as an alternative to municipal solid waste management in terms of toxicity impacts.

This becomes even more interesting when evaluating the contribution of pesticide production and application to toxicity impacts. Pesticides were a much higher contributor to toxicity impacts in the U.S. production than UK production, particularly for the ecotoxicity impacts (terrestrial, freshwater, and marine ecotoxicity) versus. the human-related toxicity impacts (i.e., human carcinogenic and non-carcinogenic toxicity). Specifically, for both U.S. tomato and kale production, pesticide production and application contributed to 50-84% of the ecotoxicity impact categories, as well as to 14-20% for human non-carcinogenic toxicity and 3-5% for carcinogenic toxicity. However, for UK kale production, pesticide production and application contributed to only 1-7% of ecotoxicity categories and <2% to the human toxicity categories; for UK tomato production, which only used biological pesticide inputs, this contributed to ≤1% of all toxicity categories. This resulted in UK conventional production generally having lower toxicity impacts than U.S. conventional production, on average. The differences between the contribution of pesticides to toxicity impacts for U.S. and UK field-based production can be attributed in part to rates of pesticide application, in kg active ingredient per area, which are an order of magnitude higher for U.S. than UK farms. However, this may also be because of the use of pesticide active ingredients with relatively higher toxicities in the U.S., which have are not allowed use in the UK, such as chlorothalonil (HSE, 2022).

Although toxicity impacts for U.S. conventional farms were higher than UK counterparts, they were generally lower when compared to U.S. organic farms, the opposite of what would be expected. van der Werf, Knudsen and Cederberg (2020) has highlighted methodological issues in appropriating toxicity impacts from pesticides in LCAs. In particular, this study suggests that it may take 20-30 years to fully understand and evaluate toxicity impacts from pesticide use, and thus, true toxicity impacts of recently developed active ingredients would not be characterised within current LCIA methods. Indeed, De Backer *et al.* (2009) similarly identified the lack of defined characterisation factors for a host of pesticide active ingredients within LCIA methods. At the same time, LCA models are based on the average performance of processes, and thus do not capture risks for rare events or specific groups (Hauschild, 2017); in terms of pesticide toxicity, this means that LCA does not capture specific instances of heightened toxic exposure, such as for agricultural workers and their families, despite the fact that these groups have some of the highest risks to adverse health effects from pesticide exposure (Jaga and Dharmani, 2005; Kennard and Vagnoni, 2021). These exposures may also be enhanced in enclosed environments, such as glasshouses, an issue that is also not considered (Boulard *et al.*, 2011).

Although the actual health risks from pesticide exposure are highly uncertain and vehemently argued among the public and academics alike (Kennard and Vagnoni, 2021), the potential risks have been widely documented in literature and thus suggest that a more in-depth modelling of pesticide exposure pathways and of both active and inert ingredients in pesticides (the latter of which might also be toxic) is required within LCAs. Organic agriculture responds to the lack of certainty about impacts of pesticide toxicity by employing precautionary principle – i.e., not using synthetic pesticides at all. However, the benefits of this are also not captured within LCA (van der Werf, Knudsen and Cederberg, 2020).

For further detail on potential health impacts from pesticides and the pathways in which this occurs, the author suggests referring to a parliamentary briefing by Kennard and Vagnoni (2021). This work was completed by the author as part of a PhD fellowship with the Parliamentary Office of Science & Technology (Kennard and Vagnoni, 2021); a brief summary of this project is included in Appendix C for reference.

3.2.4.3 Other environmental, social, and economic trade-offs

There are a variety of other ecosystem services that can arise from agricultural production, which are not consistently modelled within LCA frameworks (Tuomisto *et al.*, 2012; Boone *et al.*, 2019). This has been particularly highlighted as one of the main issues in evaluating organic and agro-ecological farming systems through LCA (Boone *et al.*, 2019; van der Werf, Knudsen and Cederberg, 2020). Organic and agro-ecological farms are characterised by multifunctionality (Boone *et al.*, 2019), aiming to produce food, but also positively contribute to ecosystem and human health (IFOAM, 2020). In contrast, the 'sustainable intensification' discourse that often dominates conventional agriculture aims to maximise food production whilst minimising negative environmental impacts (The Royal Society, 2009; Godfray *et al.*, 2010). As LCA tends to focus on assessing negative impacts in relation to product output, rather than also considering positive impacts, it can be seen that this method lends itself toward the latter case (van der Werf, Knudsen and Cederberg, 2020). Thus, many practices that may be employed by conventional and organic farmers alike to provide environmental, social, and economic benefits to their local environments and communities can be lost within LCAs. In some cases, avoided burdens, different allocation methods, and additional models have been developed to try to capture these additional ecosystem services (Venkat, 2012; Saer *et al.*, 2013; Boone *et al.*, 2019; van der Werf, Knudsen and Cederberg, 2020), but there are also many uncertainties about how to adequately quantify these services.

As compost production was a main driver of impacts for organic farms in this study, the burdens and benefits associated with this process in particular were explored. The benefit of composting as an alternative to municipal solid waste management was applied in this study by subtracting avoided burdens within Scenario 3. Additionally, the carbon sequestration benefits associated with compost was applied within a sensitivity analysis, considering a literature average for avoided CO_2, as well as a 'maximum' carbon sequestration value that also considered carbon savings from the increased water retention and reduced soil erosion associated with compost application (Saer *et al.*, 2013). When these benefits of compost were applied, many of the urban and peri-urban farms that originally had some of the highest global warming impacts due to high levels of compost application, instead had some of the lowest impacts, even negative in many cases. This implies that the high burdens of compost production and application might actually be offset by the other benefits from producing and applying. These benefits were not fully quantified; for example, the ability of compost to remediate urban soils, and thus safeguard human health (Brown and Jameton, 2000; Wortman and Lovell, 2013; Altieri and Nicholls, 2018), was not considered. On the other hand, there are also potential disadvantages that were not considered, such as the potential for compost application to actually contaminate soils (Perrin, Basset-Mens and Gabrielle, 2014), for example through microplastics (Weithmann *et al.*, 2018; Vithanage *et al.*, 2021). These represent trade-offs that need to be further explored within LCA frameworks, especially for organic farms.

The general ability of agricultural soils to sequester carbon, and thus provide a potential climate change mitigation solution (Lal, 2004; Liu *et al.*, 2006; IPCC, 2022), has also been explored in other LCAs (Meisterling, Samaras and Schweizer, 2009; Venkat, 2012). Venkat (2012) particularly considered this in relation to organic agriculture, modelling the additional carbon that can be sequestered on farms transitioning from conventional to organic production. This provided an avenue to consider some of the benefits of organic agriculture within an LCA framework, particularly the increased levels of soil carbon and organic matter commonly seen in organically-farmed soils (Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012). However, modelling soil carbon storage is an uncertainty within agricultural LCAs, due to both the lack of a standardised modelling process as well as the lack of certainty around the benefits and trade-offs of practices to increase soil C sequestration. For example, it is not an organic certification that leads to higher levels of soil carbon, but rather specific management practices that are commonly used on organic farms (e.g., applying high levels of organic matter), but can also be applied on conventional farms.

There are also disadvantages to certain organic practices that need to be considered; for example, higher application of organic fertilisers could also result in higher N_2O emissions, thus negating any carbon sequestration benefits (IPCC, 2022). This was seen within this study when average carbon sequestration benefits from compost application were considered; this slightly lowered global warming potentials for organic farm category averages (changes of ≤27%), but farms applying the highest levels of compost still had the highest impacts. Converting to organic can also result in lower yields, which, if done on a large-scale, would require additional land to be converted to cropland (Muller, Schader, *et al.*, 2017). The emissions associated with this land use change could again negate any C sequestration benefits (Leifeld *et al.*, 2013; IPCC, 2022), and could be explored more through consequential LCAs. It thus becomes obvious that modelling the trade-offs and carbon storage potential of different farming systems or farming practices is a complicated matter that requires additional evidence and standardised practice to be incorporated into LCA.

Besides carbon sequestration, farming systems can also contribute to other environmental services such as soil health and biodiversity. These services are generally not integrated into the main impact assessment methods used within agricultural LCAs (van der Werf, Knudsen and Cederberg, 2020), although soil quality and biodiversity indicator scores have been developed within the SALCA method (Oberholzer *et al.*, 2006; Jeanneret *et al.*, 2014). Many farms in this study, organic and conventional and alike, engaged in practices that contributed to these soil health and biodiversity, such as utilising min- or no-till practices and maintaining soil cover through cover crops (Palm *et al.*, 2014). Additionally, many of the organic farms had extensive areas of hedgerows or woodlands, which serve as important habitats (Heath *et al.*, 2017; Alignier, Uroy and Aviron, 2020). Certain conventional farms, especially in the

UK, also had hedgerows as well as field margin areas left to natural vegetation or seeded with wildflower or birdseed mixes. The fact that these areas that were left to 'nature' existed on many farms (especially organic ones) supports Boone *et al.* (2019)'s argument that environmental impacts from LCAs should be allocated among the ecosystems services supplied by the farm as a whole rather than specified to a sole crop output, thus considering farm multi-functionality. This would also allow for the trade-offs that exist between different land uses, environmental impacts, and crop productivity to be explored (Hodgson *et al.*, 2010).

Aside from other environmental impacts, there are also social and economic factors that are not generally accounted for within LCAs. For example, Blanke and Burdick (2005) highlighted additional benefits that local agriculture can provide, such as preserving the countryside, providing local employment opportunities, and public engagement. Although economics can be explored through LCA by assessing impacts based on prices or profit (e.g., Hu *et al.*, 2019), it may be more difficult to explore how farms contribute to local economies. In addition, this evaluation would also require a justice lens, i.e., examining the quality and working conditions of the jobs provided. A trade-off may present itself in terms of quality and quantity; large-scale farms provide a higher number of local jobs by employing farm workers, harvesters, packhouse employees, and potentially drivers; on the other hand, many organic farms in this study had few employees (generally 1-5, if at all), but often provided teaching and training opportunities through apprenticeship programmes.

From a social aspect, it is important to consider how farms contribute to local food security and improve local food access. For example, some of the conventional and organic farms in this study donated unsold produce to local churches, food banks, and social food projects, thus improving access to local produce for potentially marginalised groups. Additionally, many of the U.S. organic farms participated in the 'Georgia Fresh for Less' programme, which allows Supplemental Nutrient Assistance Program (SNAP) recipients to double their SNAP dollars (formerly known as food stamps) when used at partnered farmers' markets or on-farm markets (Wholesome Wave, 2022). This can make organic or local produce, often considered as more expensive than conventional items in supermarkets (Donaher and Lynes, 2017; Salisbury *et al.*, 2018; Siegner, Sowerwine and Acey, 2018), more affordable and accessible for low-income individuals. Some organic farms also offered discounted or free produce to volunteers as a way to make their food more accessible for those who may not have financial resources. While larger conventional farms typically produce more food, small-scale farms can also contribute to food security in their local communities through these types of initiatives. However, it's important to recognise that these social and economic aspects of farming may come at a trade-off in terms of productivity or other environmental impacts. The field of 'social or socio-economic LCAs' is working to quantify these types of trade-offs within LCA frameworks, but this is still in its early stages (Frank, Laginess and Schöneboom, 2020).

3.2.5 Ideal models and recommendations

In this section, recommendations are provided for the ideal production models for farm small-scale farms, based on the impacts assessed within this LCA, as well as the other trade-offs that have been discussed. Much of this depends on specific local context, so care should be taken not to widely generalise these recommendations; still, when examining individual farm operations, certain trends have emerged.

In particular, low global warming, fine particulate matter formation, acidification, and eutrophication impacts were seen on organic farms generally in two cases: 1) in cases where very low inputs were applied (e.g., only minor compost or manure additions, or cover crops), which also generally resulted in very low yields; and 2) in cases where moderate levels of inputs were applied, but with intensive production that resulted in high yields. In the latter case, land use impacts were also generally low due to the high yields; thus, this case likely provides a more ideal model for the small-scale, organic farm, as this provides benefits both for food security (in terms of more food produced) and relatively low environmental impacts. However, there also may be many other ecosystem and social services provided on low-input farms that have not been captured in this study, which requires further investigation.

The ideal scale that these low impacts with high productivity is achieved on differs between countries. In the U.S., the lowest impacts were seen on peri-urban farms. This type of production on the urban fringes was found to work well because transport burdens were low or nil compared to rural organic and conventional farms. Additionally, these low-impacting farms used markedly less compost in comparison to the urban farm, where high compost use was seen to be a main driver of impacts. This suggests that perhaps better quality soils were available on the urban fringes, negating the need for high compost use. Moderate levels of compost inputs or other fertilisers could thus still produce moderate to high yields. Additionally, these farms were generally very small scale (0.1-0.25 cultivated acres), employing labour-intensive growing strategies that resulted in low harvest and post-harvest waste and high yields. However, just being a 'peri-urban' farm did not necessarily result in low impacts – indeed, some of the U.S. peri-urban farms also had the highest impacts, particularly those that used relatively high levels of compost use and also had high transport burdens. This suggests that the peri-urban model is not ubiquitously low-impacting, but that it can be if resources are used efficiently, and produce is sold nearby.

For the UK, the lowest impacts among organic farms were seen for the two rural organic farms. These farms generally produced some of the highest yields, thus showing efficient resource use. Both farms used manure as a fertility input, which provided impact benefits in that production or storage emissions were not attributed to manure. The use of manure on these farms, ideal due to their more rural location, provides an opportunity for waste cycling both within and between farms. Out of these two rural organic farms, one was able to produce on a larger field scale (27 cultivated acres in total), thus achieving economies of scale in terms of infrastructure and cultivation equipment; the other produced on a more intensive, smaller scale, growing all crops in polytunnels in relatively short growing seasons. This allowed for greater output throughout the year, which led to this farm achieving the highest yields out of UK organic farms for both crops. Additionally, although these farms were rurally located, they were still located close enough to their final points of sale such that transport burdens were not a major contributor to impacts. This is opposite to what was seen in the U.S., where the farther location of the rural organic farms from urban areas and the low to moderate yields achieved on these farms negated any benefits from economies of proximity or economies of scale, despite the larger size of these farms in comparison to the other organic farms studied.

Urban agriculture in both countries was characterised by relatively high resource use, generally from compost inputs. In the U.S., these high inputs also resulted in the urban farm having some of the highest yields out of all other farms, including conventional farms; this is consistent with McDougall, Kristiansen and Rader (2018)'s study in Sydney, Australia, which also found high yields in urban gardens and farms but also with high levels of inputs. In the UK, however, the urban farms had some of the lowest yields. This suggests that more resource-efficient methods are required for urban production, which may only be suitable if appropriate land with good quality soil is available. Of course, when benefits to compost were applied in the form of avoided burdens of the municipal solid waste stream, or from the maximum carbon sequestration value, these urban farms had some of the lowest impacts. This suggests that city planners should consider urban farms as a potential avenue for the generation of waste cycling and circular economies within cities and for soil regeneration (Pearson, Pearson and Pearson, 2010; Kennard and Bamford, 2020; Rufí-Salís *et al.*, 2021); if LCA is used to evaluate urban agriculture, the specific services prioritised by the city should be considered so that the benefits of these farms can also be evaluated, not just the negative impacts. Additionally, the high yields achieved by the U.S. urban farm provides further evidence that the use of small-scale, labour-intensive growing methods can result in high production outputs and that these farms can be important sources of fresh produce in cities (Kennard and Bamford, 2020).

Finally, conventional farms in the U.S. were also seen to achieve relatively low impacts and high yields across the board, as well as certain conventional kale farms in the UK. This suggests the importance of economies of scale in reducing impacts on these farms. Thus, it is clear that low impacts can be achieved both on small-scale, as well as larger-scale farms; without a comprehensive assessment of other ecosystem services provided, one does not necessarily outweigh or emerge as the ideal over the other. Thus, this research suggests that both can play a role in contributing to food security and food system sustainability from the perspective of the environmental impacts evaluated in this LCA, but that a more robust assessment of the other environmental, social, and economic services and disservices that these farms provide is required.

3.2.6 Study limitations

Although this study presents one of the most (if not the most) comprehensive lifecycle assessments on different types of local agriculture to date, there are obvious limitations, many of which have already been discussed. In terms of data reliability and uncertainties, this study relied mainly on information provided by farmers to model the types and amounts of resources, infrastructure, equipment, materials, and other inputs used. Thus, uncertainties in modelled amounts depend highly on the level of detail, methods of tracking, and reliability of the farmers' own data. Secondary data sources, mainly ecoinvent libraries, were then used to model the production of these resources, which obviously presents another degree of uncertainty as these are global datasets that do not necessarily represent the specific materials used in the UK or Georgia, USA. However, although uncertainties exist, the primary and secondary datasets used were the most reliable data sources available at the time, within the time and resource constraints of this research study. Future improvements could be made, such as setting up universal data tracking systems across case study farms to ensure that lifecycle inventory data is collected and reported in the same way, but this would require either additional time from participating farmers or additional resources for researchers.

Additionally, although this study comprises one of the largest amounts of case study farms assessed within a singular LCA, this research would have been improved by including even more farms. In particular, the ideal would be to include more than one farm for every farm category, which was mainly an issue for urban and conventional farm categories. The difficulty with this for the prior was the smaller number of commercial urban farms that exist in general, compared to other farm categories. For the latter, there was difficulty in recruiting farmers in this study within the U.S. Also, due to the data-protective nature of the UK conventional tomato production industry, it was not possible to obtain detailed information for all inputs; this creates another degree of uncertainty, as in many cases, literature data was used to supplement available information. Finally, this study would also have been improved by being able to assess other farm categories that have not been included. Small-scale conventional farms in urban, peri-urban, and rural contexts would have provided an additional point of comparison, but these farms were difficult to find, and did not appear to exist especially within urban and peri-urban contexts. Additionally, the inclusion of larger-scale organic operations, especially those that sold into traditional supermarkets, would have provided another interesting point of comparison. It is recommended that this be explored in further research comparing scales of organic agriculture.

Finally, although this research provides important contributions to assessments of local agriculture sustainability, it also focuses on a limited set of environmental impacts and crop lifecycles. As has been discussed, additional assessments considering other ecosystem services would improve the comparisons made between organic and conventional agriculture and allow for the multifunctionality of these systems to be explored (Boone *et al.*, 2019; van der Werf, Knudsen and Cederberg, 2020). Although it was not possible within the resource and time constraints of this research, as well as of the participating farms, the study could also be improved by evaluating farms as a whole (Perrin, Basset-Mens and Gabrielle, 2014). This could be achieved by examining whole crop rotations or total farm outputs, linked with associated ecosystem services. In particular, this type of system expansion could help capture some of the co-production and waste cycling that occurred on the farms (e.g., between animals and crops), especially seen on the organic farms. However, it would be necessary to develop additional models in order to make comparisons between collective sets of food products and functions produced across farms.

3.2.7 Critique of LCA methods

The many functions and services that are not captured within LCA have already been discussed; thus, in this final section, the methodologies used within LCA frameworks are critiqued.

Although the ISO 14040 and 14044 standards (ISO, 2006a, 2006b) provide a basic guidance and framework to perform an LCA, the particular methodologies and practices used are not consistent across the field. It is left up to the LCA practitioner to select which items to include (e.g., inclusion of capital goods and infrastructure), where to set system boundaries, which emission models to use, and how to allocate burdens for co-products and by-products. For agricultural LCAs in particular, the types of emissions models used to calculate direct agricultural emissions and the types of allocation methods used have been found to critically influence final results (Meisterling, Samaras and Schweizer, 2009; Perrin, Basset-Mens and Gabrielle, 2014; Poore and Nemecek, 2018a). This was also identified within this study, particularly with respect to nitrate leaching models, which were explored within sensitivity analyses, and the methods of allocation for compost production, which were explored within three different scenarios. Both of these were found to critically influence final results and change the overarching trends observed between farm categories.

3.2.7.1 Emission models

The calculation of direct agricultural emissions is a major component of agricultural LCAs, and typically this relies on various emission models rather than direct measurements. However, a major issue is the unsuitability of these emission models for local contexts or certain types of production. For example, many studies use generalised IPCC emission factors (IPCC, 2006c, 2019), which do not account for specific local conditions or certain management practices that can influence gaseous emissions, such as rates of tillage (Venkat, 2012; Avadí *et al.*, 2022). However, there is a trade-off between the amount of data, time, and expertise required to use more complex, site-dependent emission models, versus the more generic emissions factors such as those provided by the IPCC (Avadí *et al.*, 2022).

Further, many agricultural emission models are based on rates of N inputs, rather than accounting for the actual amount of N left in the system (Bos *et al.*, 2014). Additionally, many models are based on assumptions from conventional agriculture, and thus are not adapted to suite the mode of action of organic fertilisers, particularly slow N release (Meier *et al.*, 2015). Indeed, in a field experiment measuring nitrate leaching rates from slurry and mineral fertiliser application, Frick *et al.* (2022) found that leaching from manure or even mineral fertiliser continued for 2-3 years after application and ranged between 3-10% of applied N (Frick *et al.*, 2022). For compost, it is estimated that 3-5% of N becomes available in soluble forms during the 2-5 years after application (Sullivan *et al.*, 2018). Thus, the actual amounts of N that can contribute to N-related emissions depend upon previously applied fertilisers as well as those applied the current year. For soils on organic farms in particular, which generally have relatively high levels of organic matter (Tuomisto *et al.*, 2012), organic N mineralisation can be an important factor for nitrate leaching (Frick *et al.*, 2022); this presents a degree of uncertainty in LCAs (including this one), as this often is not taken into account due to limited data on soil characteristics (Brockmann, Pradel and Hélias, 2018). Thus, it becomes clear that the most appropriate way to estimate N-related emissions would be to account for prior additions of N fertilisers, especially organic ones, and estimate the amounts of N that might become available through mineralisation each year (Venkat, 2012); however, this is difficult to account for, as it would require detailed data on fertiliser application history for each field or plot, which may not be available.

The modelling of nitrate leaching is a particularly important challenge in LCAs, as it has been found to be one of the most important N emission pathways (Pardon *et al.*, 2016), whilst also being one of the emissions most sensitive to model selection (Henryson *et al.*, 2020; Avadí *et al.*, 2022). This is why five nitrate leaching models were explored as a sensitivity analysis within this study. The selection of model was not found to influence results for global warming impacts but did affect trends observed between marine eutrophication impacts. The generic IPCC 2019 Tier 1 emission factor (IPCC, 2019) was generally (but not always) found to generate the highest nitrate leaching emissions in comparison to other site-dependent nitrate leaching models. However, large variance in marine eutrophication impacts was also observed between the site-dependent models evaluated, which included the de Willigen 2000 model (de Willigen, 2000), as the default method, and also the SQCB-NO_3 model (Emmenegger, Reinhard and Zah, 2009), the Smaling 1993 model (Smaling, Stoorvogel and Windmeijer, 1993), and the Poore-Nemecek model (Poore and Nemecek, 2018b). Still, the lowest marine eutrophication impacts on organic farms were generally seen when calculating nitrate leaching based on soluble N instead of total N generally (using the default model), thus highlighting the importance of developing nitrate leaching models that better account for N fluxes in organic systems (Bos *et al.*, 2014; Meier *et al.*, 2015).

In summary, the choice of emission models can significantly impact the results of agricultural LCAs, potentially altering comparisons made between systems. To address this issue and reduce the potential for bias or subjectivity among LCA practitioners, it is necessary to establish standardised methods for selecting emission models in agricultural LCAs.

3.2.7.2 Allocation and avoided burdens

Methods of allocation are one of the most hotly contested topics within LCA, with a wide range of methods employed that are ultimately up to the LCA practitioner (Ardente and Cellura, 2012). Despite ISO 14044 guidelines suggesting allocation methods in order of preference and also suggesting that LCA practitioners conduct a sensitivity analysis to explore various allocation methods whenever several are applicable (ISO, 2006b), these suggestions are not consistently applied across LCAs. However, the selection of allocation methods has been identified as one of the most important determinants of results in agricultural LCAs (Perrin, Basset-Mens and Gabrielle, 2014). In particular, avoided burdens or other positive impacts (e.g., carbon sequestration) tend to be inconsistently applied among LCAs and produce varied effects on results (Dorr *et al.*, 2021).

This LCA particularly explored various allocation scenarios for the production of compost and for the use of CHP, the latter of which has been previously explored in more detail in the context of agricultural systems (Blonk *et al.*, 2010; Antón *et al.*, 2012; Torrellas, Antón, Ruijs, *et al.*, 2012). Global warming, fine particulate matter formation, terrestrial acidification, and marine eutrophication impacts were particularly sensitive to different methods of compost allocation. As compost production burdens were one of the main driving factors for impacts on organic farms, when these burdens were not included in Scenario 2, this dissipated much of the impact variance observed between these farms. The application of avoided burdens in Scenario 3 completely shifted the trends observed for global warming and marine eutrophication, where farms applying the highest levels of compost were now seen to have overall negative impacts in many cases. If Scenario 3 had been considered as the default scenario, this would have resulted in an entirely different set of overall conclusions, where mainly urban and peri-urban farms that applied high levels of compost would actually emerge as the lowest-impacting models (at least for global warming and marine eutrophication impacts).

This shows how sensitive LCA results are to the allocation methods applied and the positive impacts or avoided burdens considered. This can create uncertainty about the validity and reliability of LCA as a method to compare the environmental sustainability of different products or systems and thus to influence decision-making in business or in policy settings. To address this issue, it is imperative that LCA practitioners are transparent about the allocation methods and benefits applied in their analyses. In accordance with ISO 14044 standards, LCA results should be provided for a range of allocation scenarios to show the sensitivity of the results to choices made by the practitioner. In addition, when considering avoided burdens or other applied benefits (such as carbon sequestration), LCA results should also be provided for scenarios without these benefits to ensure objectivity and avoid skewing the results towards the subjective biases of the practitioner or funder. This is crucial to ensure that LCA results are objective and reliable, and to avoid drawing system boundaries or applying benefits in ways that could distort results.

3.3 Chapter conclusion

Overall, this study showed that the specific management decisions practices employed on a farm are the most important in determining environmental impacts, rather than broad designations of 'organic vs. conventional' or local scale. This study showed that LCA results are highly sensitive to the choices made by the LCA practitioner, such as the emission models and allocation methods used. Thus, caution should be exercised when interpreting and applying results from LCAs, particularly in policy settings. It is imperative that more standardised procedures are developed for agricultural LCAs, especially for calculating emissions and accounting for burdens of waste resources like compost. While LCA is a powerful tool for comparing farming systems and identifying impact hotspots, it should not be used to draw wide-ranging conclusions about the most 'sustainable' farming systems, as this can depend on a variety of factors such as the individual farms studied, the crops examined, and the geographical context. To truly understand farm sustainability on a systems level, it is necessary to also consider the positive contributions that farms can make to their local communities, environments, and economies, including the ecosystem services they provide, which are typically lost within LCAs. Thus, these ecosystem services should be evaluated in addition to traditional LCA frameworks, which is the aim of the following chapter.

Chapter 4: Soil health and ecosystem services from local farms in the UK

This Chapter builds upon the evaluation of environmental impacts in Chapter 3 by exploring the additional ecosystem services and disservices that may be provisioned on these farms. Key soil health indicators were tested across ten of the UK-based farms that participated in the LCA study; the results from this assessment are presented and discussed within this Chapter to provide insight to potential soil-based ecosystem service provisioning. More detail on the methodology employed for this study has been provided in Section 1.1.

4.1 Results

4.1.1 Principal component analysis (PCA)

Principal component analysis (PCA) was first used to identify experimental variables and grouping parameters of interest in order to reduce the high-dimensional dataset for further analysis. This was performed on all measured soil parameters, as well as all numerical characterising information (e.g., temperature, rainfall, latitude, longitude, etc.) without separation of data into groups. Measured soil parameters at each depth (0-10 cm as the 'upper' depth and 10-20cm as 'lower' depth) were considered as separate variables within this model.

Results from the PCA are presented in Table 64, which includes the eigenvalues and proportion of variance for each PC. Six PCs accounted for 64% of the variance. Figure 66-A provides a further visualisation of the eigenvalues for each PC as a scree plot. Table 65 then provides the factor loading scores (eigenvector weight) from each variable for the first six PCs. Listed variables are designated with a 'u' or 'l' in parentheses for those sampled at upper or lower depths, respectively.

PC scores and loadings from the analysis were examined via biplots to uncover the relationship between samples and variables. The biplot for PC1 and PC2, which collectively account for 33.5% of variance, is provided in Figure 66-B. The biplot highlights the main variables influencing variance in each direction, with variables in the plot listed in approximate descending order of vector magnitude based on factor loadings for PC1 and PC2. Factor loadings between upper and lower depth variables were often similar, so these are not differentiated within Figure 66-B unless major differences arose between depth (i.e., different directions or magnitude of loadings).

Table 65 and Figure 66-B were used to identify the highly weighted variables that drive the majority of variance in the dataset and also to examine correlation between variables. PC1 accounted for 19.6% of variance and showed high positive loadings ($\geq$0.8 loading value) from Fe, As, Co, Mn, and Cu (in descending order), as well as a smaller positive loading (approximately 0.3) from Zn in the same direction. It can thus be seen that many of the tested heavy metals were correlated to each other and associated mainly with PC1. PC1 also had a high negative loading from longitude (-0.60).

PC2 accounted for 13.9% of variance, with relatively high positive loadings (>0.5) from Mo, Pb, latitude, and water-holding capacity (WHC), in descending order, with smaller loadings from P (>0.35), Cd (>0.3), SOC ($\geq$0.1), and N ($\geq$0.1) in approximately the same direction.

PC2 also had high negative loadings (≤-0.50) from bulk density (BD) and temperature-related variables. Finally, farm size and Mg were associated with both PC1 and PC2, providing positive loadings ≥0.45 for PC1 and negative loadings <-0.40 for PC2.

PC3 accounted for 11% of variance and showed high negative loadings (≤-0.60) from other elemental variables that were not seen as main contributing factors for PC1 and PC2, including K, B, and Ca. However, these elements were not identified to be of particularly high interest in relation to ecosystem service provisioning, so they were not selected for further analysis. PC4-6 accounted for <8% of variance individually. For these PCs, no other variables emerged with an absolute value factor loading >0.6, and the variables with relatively higher weightings for each of these PCs had already been identified as important factors for PC1 and PC2. Thus, further investigation of PC scores within the PCA model focuses on PC1 and PC2.

Out of the numerical characterising variables, latitude, longitude, temperature, and farm size had the highest loading factors for PC1 and PC2, each influencing variance in different directions. Longitude is negatively correlated with farm size, Mg, and S. Temperature was positively correlated with bulk density and negative correlated with latitude, N, and P. This suggests the potential importance of temperature and location-related variables in driving variance in macronutrient soil concentration and bulk density; however, these variables do not appear to be correlated with most heavy metal variables.

Based on the main variables identified as influencing the highest amounts of variation for PC1 and PC2, as well as those that are of most interest in terms of soil health and ecosystem service provisioning, a subset of variables was selected for further analysis. These are designated in boldface in the biplot (Figure 66-B). In particular, metals such as Fe, As, Cu, Pb, and Cd were selected because each had high influence on either PC1 or PC2; in addition, the prior three and the latter two represent high loadings in nearly orthogonal directions, indicating that these groups of metals drive variance within the data in an uncorrelated manner, and thus highlighting the need for further exploration. Although there were other variables that also had high loadings in similar directions to these metals, such as Mo, Co, and Mn, these were not selected because it was assumed that they would behave in a correlated manner, and the other metals were identified to be of more interest for this study.

Water-holding capacity (WHC) was also selected, as this had a high loading factor in a similar direction to Pb and Cd, but provides a distinct variable related to soil health in addition to the heavy metals. Bulk density (BD) was selected as a variable with a high loading value for PC2, which appeared to be negatively correlated to WHC and other variables in a similar direction. Finally, certain variables were selected for further analysis even though they had relatively small loadings for PC1 and PC2, including soil organic carbon (SOC), N, C/N ratio, and P. These variables were still included because of their relevance to soil health and ecosystem service provisioning.

Table 64 – *Results of PCA for soil health assessment*

	PC1	PC2	PC3	PC4	PC5	PC6
Eigenvalue	11.16	7.90	6.24	4.33	3.60	3.27
Proportion of variance	19.58%	13.87%	10.95%	7.60%	6.31%	5.73%
Cumulative proportion of variance	19.58%	33.45%	44.40%	52.00%	58.31%	64.05%

Table 65 – PCA factor loadings for soil health assessment

Variable	PC1	PC2	PC3	PC4	PC5	PC6
Latitude	-0.29	0.64	-0.13	0.41	-0.10	0.41
Longitude	-0.60	0.26	-0.36	-0.22	0.26	0.46
Elevation	0.14	0.29	0.49	-0.54	0.34	0.06
Rainfall	0.23	-0.11	0.11	0.34	-0.56	-0.46
Minimum annual temperature	0.08	-0.78	-0.17	-0.30	0.08	-0.42
Maximum annual temperature	-0.08	-0.71	-0.19	-0.44	0.36	-0.23
Mean annual temperature	-0.01	-0.76	-0.19	-0.40	0.25	-0.32
Annual sunshine duration	0.23	-0.02	0.06	-0.46	0.52	0.31
Farm size	0.72	-0.56	0.06	0.02	-0.04	-2.0E-03
BD (u)	-0.18	-0.54	-0.11	-0.27	-0.51	0.28
BD (l)	-0.16	-0.56	-0.07	-0.34	-0.53	0.15
WHC (u)	0.20	0.51	-0.11	0.19	0.28	-0.31
WHC (l)	0.23	0.59	0.01	0.28	0.20	-0.17
C (u)	0.17	0.24	0.18	0.17	0.39	-0.37
C (l)	0.25	0.09	0.07	0.35	0.34	-0.47
N (u)	-0.02	0.13	0.12	0.20	0.06	0.10
N (l)	0.03	0.33	-0.07	0.02	-1.1E-03	0.23
C/N (u)	-0.10	-0.33	0.31	-0.07	-0.04	-0.31
C/N (l)	-0.18	-0.31	0.21	-0.10	0.16	-0.34
S (u)	0.09	-0.12	-0.33	0.21	0.33	0.36
S (l)	0.11	-0.11	-0.30	0.21	0.33	0.35
P (u)	0.04	0.40	-0.46	-0.31	0.09	-0.32
P (l)	0.06	0.37	-0.35	-0.40	0.02	-0.24
K (u)	-0.11	-0.02	-0.79	0.21	0.14	-0.11
K (l)	-0.16	-0.06	-0.71	0.25	0.12	-0.11
Ca (u)	0.02	-0.05	-0.71	-0.34	0.20	-0.03
Ca (l)	0.17	0.05	-0.60	-0.53	0.06	0.28
Mg (u)	0.47	-0.45	-0.49	0.12	0.07	0.32
Mg (l)	0.47	-0.44	-0.49	0.08	-0.01	0.39
B (u)	-0.28	-0.12	-0.74	0.25	0.22	-0.19
B (l)	-0.24	-0.15	-0.72	0.15	0.22	-0.17
Mn (u)	0.84	0.10	0.16	-0.12	0.20	0.16
Mn (l)	0.83	0.03	0.22	-0.07	0.21	0.17
Cr (u)	0.15	0.21	-0.39	0.21	-0.25	-0.35
Cr (l)	0.77	-0.20	-0.20	0.41	0.00	0.12
Fe (u)	0.91	0.13	0.01	0.01	0.14	0.04
Fe (l)	0.89	0.04	0.02	0.12	0.15	0.05
Co (u)	0.90	-0.12	-0.07	0.16	0.07	0.15
Co (l)	0.87	-0.09	0.03	0.28	0.09	0.15
Ni (u)	0.13	0.09	-0.60	0.09	-0.33	-0.17
Ni (l)	0.12	-0.14	-0.17	0.18	-0.56	0.24
Cu (u)	0.81	-0.02	-0.04	-0.14	-0.13	-0.17
Cu (l)	0.81	-0.20	0.08	0.02	-0.13	-0.16
Zn (u)	0.26	-0.08	-0.57	-0.23	-0.27	-1.9E-04
Zn (l)	0.44	-0.09	-0.17	0.04	-0.53	0.02
As (u)	0.89	-0.21	0.13	-0.10	-0.05	-0.03
As (l)	0.89	-0.28	0.15	-0.05	-0.03	-0.07
Se (u)	0.15	0.32	-0.14	0.15	0.02	-0.14
Se (l)	0.44	0.50	-0.26	0.10	0.10	-0.06
Mo (u)	0.23	0.66	-0.28	-0.29	-0.18	0.04
Mo (l)	0.12	0.70	-0.06	0.12	-0.16	0.05
Cd (u)	0.30	0.47	-0.18	-0.46	-0.15	-0.07
Cd (l)	0.36	0.32	-0.12	-0.28	-0.06	-0.15
Hg (u)	0.17	0.44	-0.26	-0.03	-0.28	-0.25
Hg (l)	-0.05	0.06	0.27	-0.02	0.11	0.22
Pb (u)	0.29	0.63	-0.03	-0.58	-0.19	-0.02
Pb (l)	0.31	0.62	0.02	-0.59	-0.16	-2.6E-04

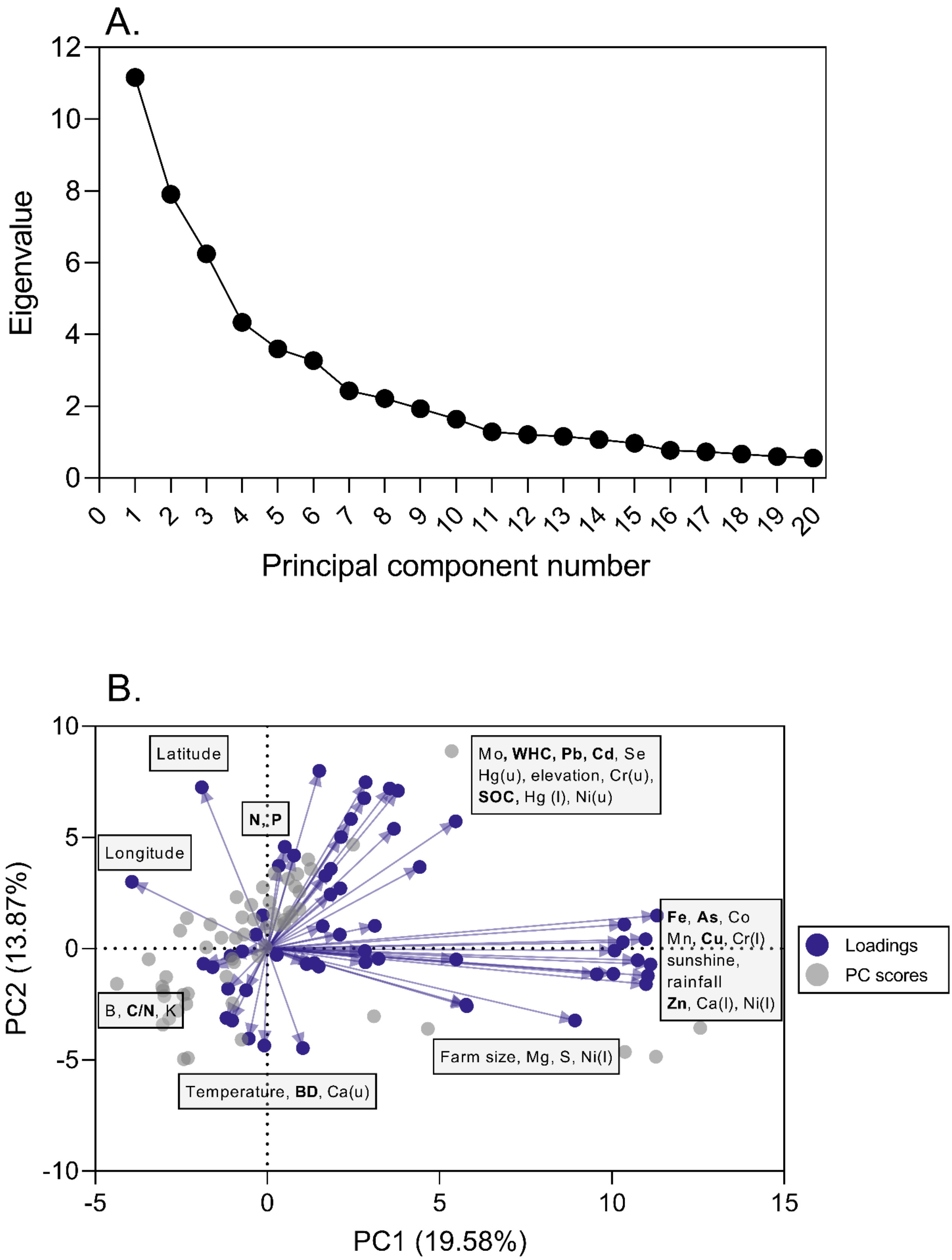

Figure 66 *– Results of PCA, with (A) presenting the scree plot (PC eigenvalues) and (B) presenting the biplot, including PC scores and factor loadings for PC1 and PC2.*

In order to identify the grouping variables that may be used to perform further statistical analysis, PC1 and PC2 scores were plotted in two dimensions and labelled in groups based on each characterising variable included in the study (both numerical and categorial). These included: latitude, longitude, farm size, annual temperature (minimum, mean, and maximum), annual rainfall, annual sunshine duration, and elevation as the numerical variables, which were also included in the PCA, and region, management type, urbanisation, landscape type, soil type, and soil parent material as the categorical variables. These plots were examined for clustering, as this indicates that statistical variation in the data may be linked to these groups.

Figure 67 presents PC1 and PC2 scores grouped based on landscape (A), management type (B), urbanisation (C), and mean annual temperature (D). Clear clusters are not observed for landscape, although this may be attributed to the relatively low number of samples per each landscape category (n=5-7). On the other hand, PC scores showed slight clustering based on management type (uncultivated, organically-managed, and conventionally-managed) and urbanisation (urbanised vs. rural soils), as portrayed in Figure 67-B and Figure 67-C, respectively. This therefore suggests that statistical variation between groups based on management type and urbanisation are likely more significant than by landscape.

Clusters of organic and urban farms appear mainly near the axis origin (Figure 67-B and Figure 67-C, respectively). Organic and urban clusters are not identical but are similar because all urbanised farms are organic, although not all organic farms are urban. Most organic and uncultivated samples are roughly separated from conventional samples by PC2; this is also the case for urban vs. rural farms, indicating the importance of variables with high factor loadings in directions parallel to the PC2 axis, such as WHC, Pb, Cd, SOC, and BD (Figure 66-B). Additionally, a higher proportion of PC scores related to urban soils exist in the positive axes of PC1 in comparison to scores for rural soils, highlighting the importance of PC1 in also driving variation between these groups. This indicates the significance of various metal variables with high positive factor loadings for PC1, including Fe, As, and Cu among those selected for further analysis (Figure 66-B).

Out of the geographically-based variables analysed, mean annual temperature resulted in the most distinct clustering (Figure 67-D), with groups separated mainly by PC2. Farms with lower annual temperatures have positive PC2 scores and those with relatively higher annual temperatures have negative PC2 scores. A clear cluster of samples from farms with lower annual temperatures (designated in dark blue) is seen near the axis origin; this cluster is similar to that seen for urbanised farms in Figure 67-C. This highlights an important point, which is that most of the urbanised farms in this study are actually from northern regions with lower annual temperatures, whilst rural farms were distributed more evenly across the south, east, and north. Other PC score plots with groupings related to mean annual temperature, such as region, minimum temperature, maximum temperature, and elevation, show similar clustering, and thus have not been reproduced.

PC scores with groupings based on soil type showed what appeared to be distinct clusters, but because most samples for each soil type were only taken from one or two different farms, there is not enough data to draw a significant conclusion from this. Thus, this plot has not been reproduced here. Finally, PC scores grouped based on farm size also portrayed some distinct clustering, but this was almost identical to that seen for urban vs. rural clustering (as urban farms have smaller farm sizes), and thus this plot has not been reproduced.

Considering this analysis of grouped PC scores, management type and urbanisation were selected as the grouping variables for further statistical analysis. Mean annual temperature was not selected as it is not related to the main interest of this study, but its PC score plot helps to inform discussion points and better understand other drivers of sample variation. It is important to note that any significant differences observed between urbanised and rural soils within the next section may also be related to the regional differences between these datasets.

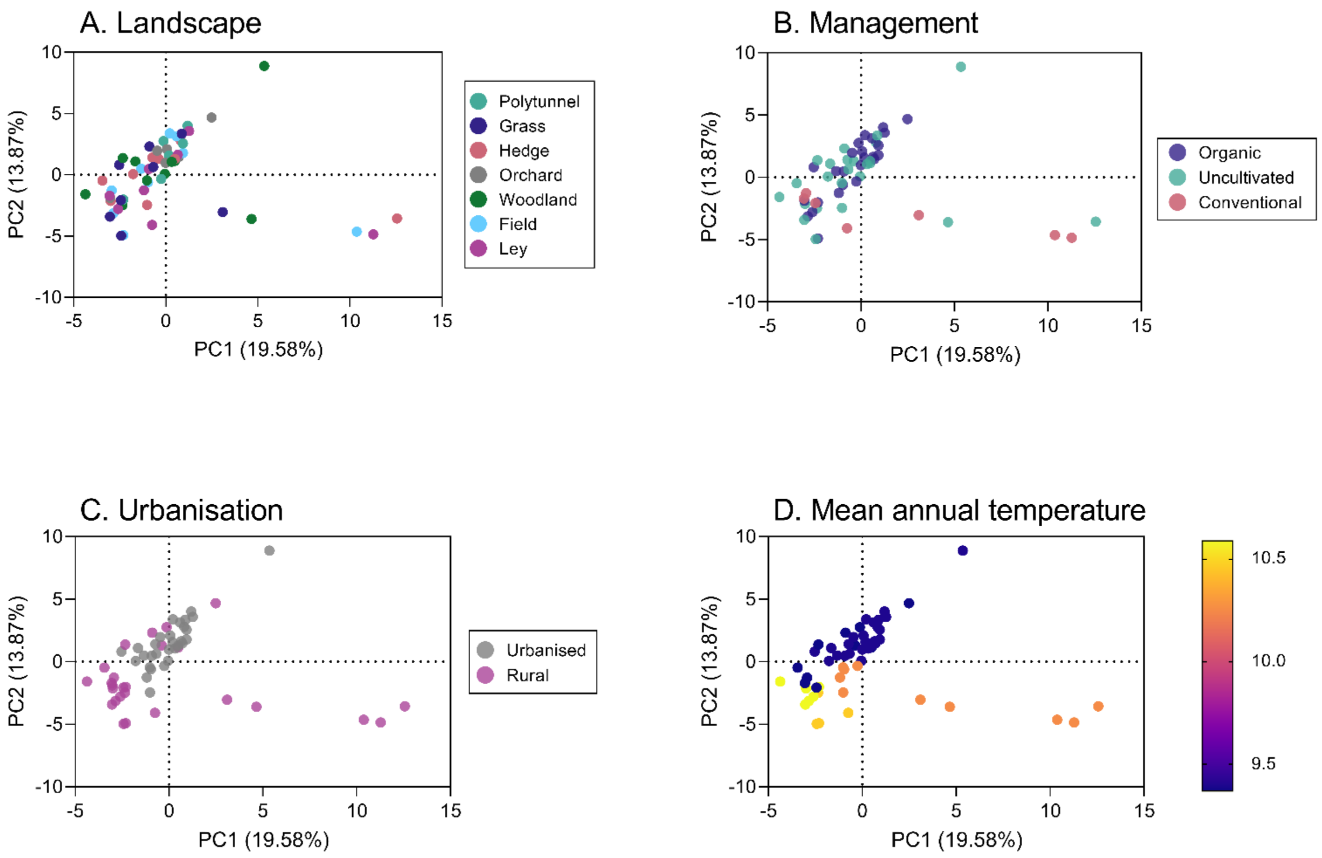

Figure 67 *– PC score plots, grouped by A) landscape, B) management type, C) urbanisation, and D) mean annual temperature.*

4.1.2 Statistical comparisons of selected soil parameters

Statistical tests were performed on the variables of interest identified by the PCA, including data from all sampled landscapes. Samples were grouped first based on urbanisation, which includes 31 soils samples collected from five urbanised farms and 27 soil samples collected from five rural farms. These were tested for differences between means using T-tests for parametric data and for differences between mean ranks using Mann-Whitney U tests for nonparametric data.

In a second analysis, samples were grouped based on management type, designated as either ‘uncultivated’, ‘organic’, or ‘conventional’. The first case refers to all land areas not used for food production (thus uncultivated), whilst the latter cases refer to cultivated land on organically-managed and conventionally-managed farms, respectively; classifications have been previously described in more detail in Section 2.2.1. This analysis considers 25 soil samples from uncultivated land areas across all ten farms; 25 soil samples from eight organic farms; and 8 soil samples from three conventional farms. It should be noted that one mainly conventional farm also has an area of organic cultivation that was sampled for inclusion within the organic category. The grouped soil samples were tested for differences between means using ANOVAs for parametric data and differences between medians using Kruskal-Wallis tests for nonparametric data.

Significant p values in the tables and figures in this section are denoted by the number of asterisks, where:

- * indicates $0.01 \le p < 0.05$
- ** indicates $0.001 \le p < 0.01$
- *** indicates $0.0001 \le p < 0.001$
- **** indicates $p < 0.0001$

4.1.2.1 Comparisons based on urbanisation

Table 66 provides a summary of the statistical tests, types of data transformation and *p* values for each soil parameter comparing soil samples from urbanised areas (n=31) or samples from rural areas (n=27), considering 56 degrees of freedom (df). Statistical test values are also provided, which include t values for T-tests, performed on parametric data, and U values for Mann-Whitney U tests, performed on nonparametric data. Tested soil parameters are designated with a (u) for those sampled at the upper depth (0-10cm) and an (l) for those sampled at the lower depth (10-20cm). Fr C/N in particular, additional statistical tests were performed due to the existence of outliers; thus, the results of these tests are also included. Detailed descriptive statistics for these soil parameters are provided in Appendix F, Table 80.

Table 66 *– Statistical tests and results for measured soil parameters from urbanised and rural soils. Soil parameters resulting in significant differences ($p<0.05$) are indicated in bold.*

Soil parameter	**Unit**	**Transformation**	**Test**	**t / U**	***p* value**
BD (u)	g cm^{-3}	None	Mann-Whitney U	317	0.119
BD (l)	g cm^{-3}	None	Unpaired T test	2.005	0.0498*
WHC (u)	%	None	Mann-Whitney U	299	0.0633
WHC (l)	%	Log	Unpaired T test	0.9569	0.3306
SOC (u)	g kg^{-1}	None	Mann-Whitney U	286	0.0390*
SOC (l)	g kg^{-1}	Sqrt	Unpaired T Test	0.4907	0.6256
N (u)	g kg^{-1}	None	Mann-Whitney U	368	0.4386
N (l)	g kg^{-1}	None	Mann-Whitney U	319	0.1234
C/N (u)	Ratio	None	Mann-Whitney U	399	0.7659
C/N (l)	Ratio	None	Mann-Whitney U	334	0.1906
C/N (u)*	Ratio	None	Mann-Whitney U	167	0.0017**
C/N (l)*	Ratio	Log	Unpaired T test	1.445	0.1552
P (u)	mg kg^{-1}	Log	Unpaired T Test	0.3941	0.6950
P_L	mg kg^{-1}	Log	Unpaired T Test	0.01438	0.9886
As (u)	mg kg^{-1}	None	Mann-Whitney U	317	0.1159
As (l)	mg kg^{-1}	None	Mann-Whitney U	292	0.0490*
Cd (u)	mg kg^{-1}	None	Mann-Whitney U	263	0.0149*
Cd (l)	mg kg^{-1}	Sqrt	Unpaired T test	4.013	0.0002***
Cu (u)	mg kg^{-1}	Log	Unpaired T test	0.2102	0.8343
Cu (l)	mg kg^{-1}	None	Mann-Whitney U	400	0.7807
Fe (u)	mg kg^{-1}	None	Mann-Whitney U	267	0.0178*
Fe (l)	mg kg^{-1}	None	Mann-Whitney U	226	0.0023**
Pb (u)	mg kg^{-1}	None	Mann-Whitney U	138	<0.0001****
Pb (l)	mg kg^{-1}	Log	Unpaired T test	5.541	<0.0001****
Zn (u)	mg kg^{-1}	None	Mann-Whitney U	226	0.0023**
Zn (l)	mg kg^{-1}	None	Mann-Whitney U	279	0.0295*

[*] Excluding outlier values (all values <1 and ≥60); df = 46.

Figure 68 then displays the mean, median, interquartile range, and data outliers for urbanised and rural groupings using box plots for the main tested soil health indicators, particularly bulk density, water-holding capacity, SOC, N, C/N ratio, and P (subplots A-F, respectively). Figure 69 provides the same information for analysed heavy metal concentrations, including As, Cd, Cu, Fe, Pb, and Zn (subplots A-F, respectively). Statistical significance between groups is designated using asterisks. Further, box plots in each figure are sub-divided with a 1 or 2 following the letter label, based on soil parameters tested on samples from the upper depth (0-10cm) and the lower depth (10-20 cm), respectively.

Few significant differences are observed between the measured soil properties and nutrient contents as portrayed in Figure 68. Average bulk density is slightly lower for urbanised farms compared to rural farms (by approximately 10% at both depths), but this difference is only significant at the lower depth. Average SOC is also 27% higher on urbanised farms at the upper depth, but no significant differences are observed at the lower depth. Additionally, no significant differences are observed between rural and urbanised soils for WHC, N, C/N ratio, or P. For C/N ratios, in particular, it can be seen that mean values are driven higher by significant outliers, mainly for rural soils. These outliers include values much higher than typically expected in these soils (C/N > 60) and occurred mainly on two farms as the result of extremely low N concentrations, rather than high C concentrations. It can be seen that these outliers shifted the C/N mean for the rural soil category considerably higher, although medians are more similar between urbanised and rural soils (Appendix F, Table 80). Considering this, additional statistical analyses were performed on this data excluding outliers (Table 66). In this case, mean and median C/N values for urban farms were significantly higher than on rural farms by approximately 46% at the upper depth, but differences were not significant at the lower depth (Appendix F, Table 80). Box plots for C/N data with removed outliers are provided in Appendix G, Figure 76.

On the other hand, significant differences between urban and rural soils are seen for many trace / heavy metal concentrations (Figure 69). Urban soils have significantly higher Cd, Pb, Zn, As (at the lower depth only), and Fe concentrations than rural soils, when considering mean concentrations for parametric data and median concentrations for non-parametric data (Table 66). The largest differences are seen for Pb and Cd, where mean concentrations in urban soils are approximately double that of rural soils. Urbanised soils also have median Zn values that are approximately 95% higher than soils at the upper depth and 63% higher at the lower depth, but mean concentrations are more similar due to the influence of outliers for the rural category. Additionally, median concentrations of Fe are approximately 50% higher on urban soils at both depths. Like for Zn, mean As concentrations are driven by many outliers, especially in the rural category, which actually sees approximately 40% higher mean values than the urban category; however, when considering median values, urbanised soils have approximately 25% higher As. For Cu, levels are similar between urban and rural soils and no significant differences are observed.

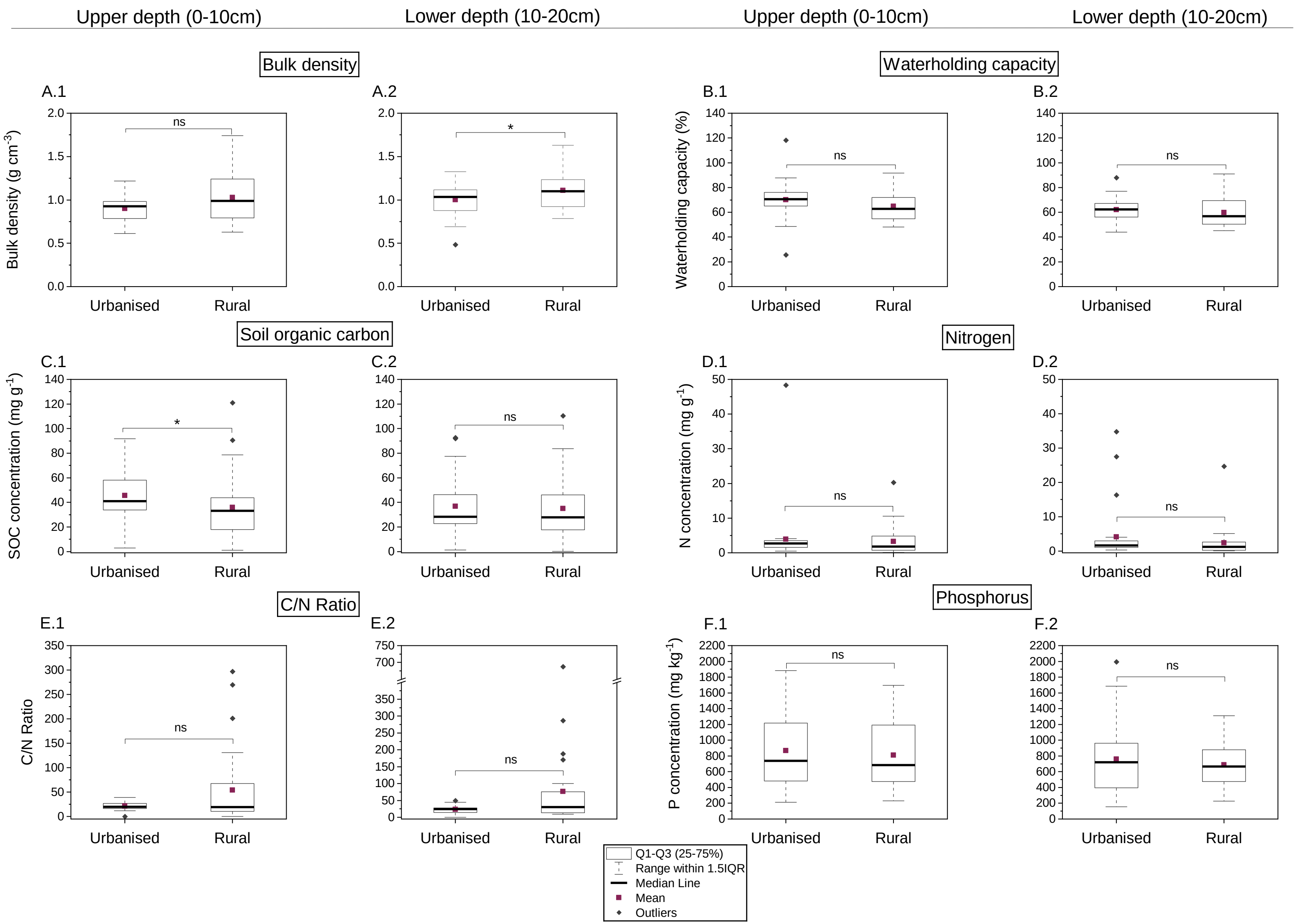

Figure 68 *– Box plots showing medians, means, interquartile ranges, and outliers for soil properties and major nutrients from urbanised and rural soils. Subplots A-F show bulk density, water-holding capacity, soil organic carbon, N concentration, C/N ratio, and P concentration, respectively. Subplots numbered with a (1) refer to soils sampled at the upper depth (0-10 cm) and (2) refer to those sampled at the lower depth (10-20cm). Significant differences between groups are indicated by asterisks; no significant differences are denoted by 'ns.'*

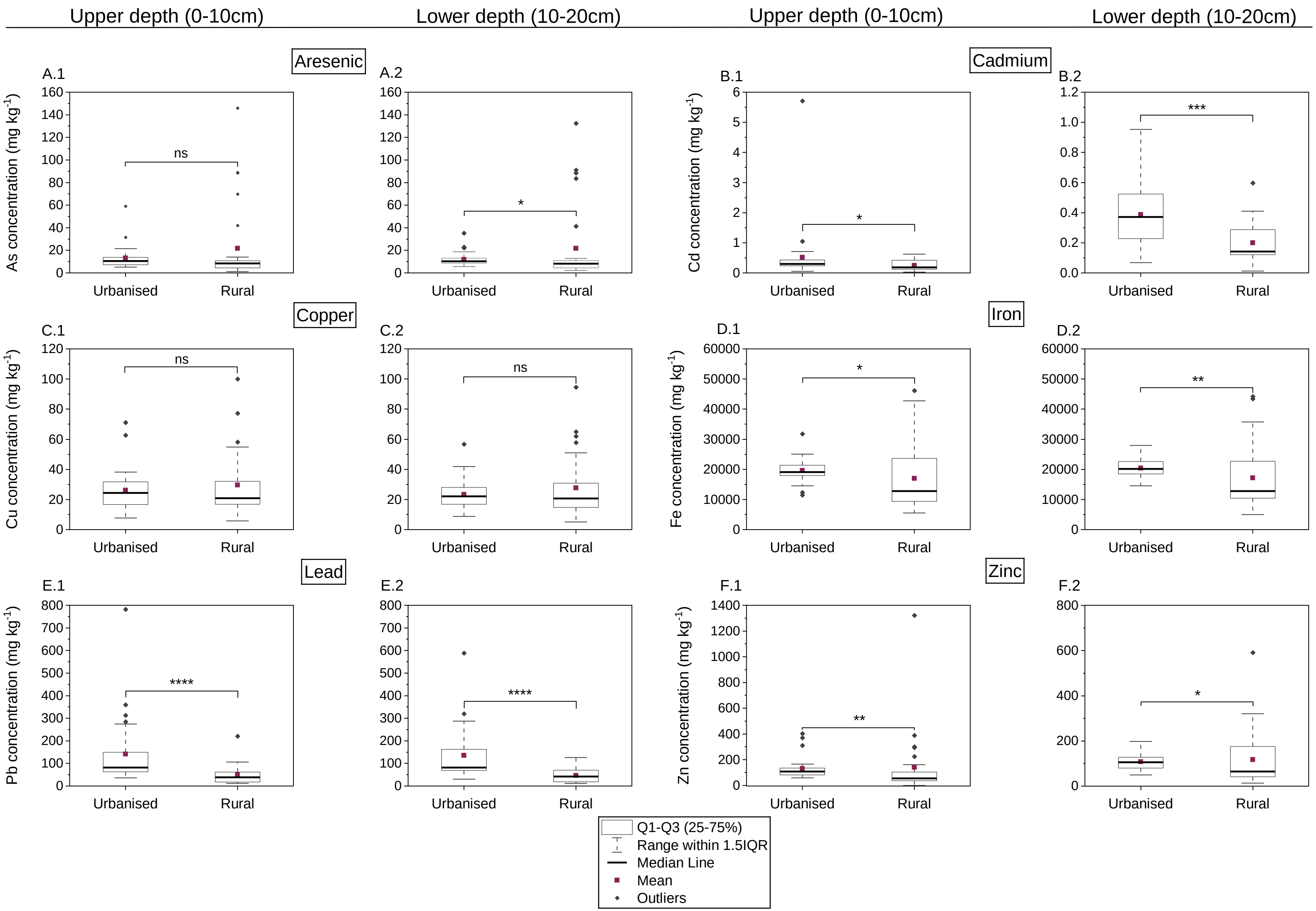

Figure 69 *– Box plots showing medians, means, interquartile ranges, and outliers for trace / heavy metal concentrations from urbanised and rural soils. Subplots A-F show concentrations of As, Cd, Cu, Fe, Pb, and Zn, bulk density, respectively. Subplots numbered with a (1) refer to soils sampled at the upper depth (0-10 cm) and (2) refer to those sampled at the lower depth (10-20cm). Significant differences between groups are indicated by asterisks; no significant differences are denoted by 'ns.'*

4.1.2.2 Comparisons based on management type

Table 67 provides a summary of the statistical tests, types of data transformation and *p* values for each soil parameter when comparing soil samples from uncultivated areas (n=25), organically-cultivated areas (n=25), and conventionally-cultivated areas (n=8), considering 57 total degrees of freedom. F values are provided for statistical comparisons made using ANOVAs for parametric data and Kruskal-Wallis test statistics (KW) for comparisons made using the Kruskal-Wallis test for nonparametric data. Additional statistical tests were performed for C/N data excluding outliers, as done when comparing based on urbanisation groupings. Detailed descriptive statistics for this data are provided in Appendix F, Table 81.

Table 67 – *Statistical tests and results for measured soil parameters from uncultivated, organic, and conventional soils. Soil parameters resulting in significant differences (p<0.05) are indicated in bold.*

Soil parameter	Unit	Transformation	Test	F / KW	*p* value
BD (u)	g cm^{-3}	Reciprocal	One-way ANOVA	10.98	<0.0001****
BD (l)	g cm^{-3}	None	One-way ANOVA	6.770	0.0024**
WHC (u)	%	None	Kruskal-Wallis	10.07	0.0065**
WHC (l)	%	None	One-way ANOVA	4.303	0.0184*
C (u)	g kg^{-1}	Square root	One-way ANOVA	6.436	0.0031**
C (l)	g kg^{-1}	Square root	One-way ANOVA	2.758	0.0722
N (u)	g kg^{-1}	None	Kruskal-Wallis	5.571	0.0617
N (l)	g kg^{-1}	None	Kruskal-Wallis	7.083	0.0290*
C/N (u)	Ratio	None	Kruskal-Wallis	2.021	0.3640
C/N (l)	Ratio	None	Kruskal-Wallis	1.193	0.5507
C/N (u)*	Ratio	Log	One-way ANOVA	0.0222	0.978
C/N (l)*	Ratio	None	One-way ANOVA	0.6859	0.5088
P (u)	mg kg^{-1}	Log	One-way ANOVA	8.066	0.0008***
P (l)	mg kg^{-1}	Log	One-way ANOVA	7.611	0.0012**
As (u)	mg kg^{-1}	None	Kruskal-Wallis	0.6229	0.7324
As (l)	mg kg^{-1}	None	Kruskal-Wallis	1.160	0.5598
Cd (u)	mg kg^{-1}	None	Kruskal-Wallis	4.232	0.1205
Cd (l)	mg kg^{-1}	None	Kruskal-Wallis	8.667	0.0131*
Cu (u)	mg kg^{-1}	Log	One-way ANOVA	2.875	0.0649
Cu (l)	mg kg^{-1}	Log	One-way ANOVA	5.027	0.0099**
Fe (u)	mg kg^{-1}	None	Kruskal-Wallis	0.5088	0.7754
Fe (l)	mg kg^{-1}	None	Kruskal-Wallis	0.4576	0.7955
Pb (u)	mg kg^{-1}	Log	One-way ANOVA	1.287	0.2842
Pb (l)	mg kg^{-1}	Log	One-way ANOVA	1.557	0.2199
Zn (u)	mg kg^{-1}	None	Kruskal-Wallis	5.942	0.0513
Zn (l)	mg kg^{-1}	None	Kruskal-Wallis	5.828	0.0543

* Excluding outlier values (all values <1 and ≥60); total df = 47.

Figure 70 and Figure 71 then display the mean, median, interquartile range, and data outliers for management types using box plots for each tested soil parameter, with significant differences between each group also highlighted using asterisk notation. Box plots are organised in a similar fashion as for urban and rural groupings.

Unlike for urban vs. rural groups, the main significant differences between management types are seen for the major soil properties and nutrients (Figure 70) rather than heavy metal concentrations (Figure 71). Organic and uncultivated soils tend to have more similar median and mean values for the main soil health indicators provided in Figure 70 compared to conventional soils. With the exception of P concentration, no significant differences between uncultivated and organically-managed soils are observed for these soil health indicators. However, organically-cultivated soils have approximately 40% higher P concentrations than both uncultivated and conventional soils at both depths, although the difference was not significant for conventional soils at the lower depth.

On the other hand, conventionally-managed field soils have significantly higher bulk density and significantly lower water holding capacity than both organic and uncultivated soils (Figure 70). For upper and lower depths respectively, average bulk density of conventional soils is approximately 30% and 20% higher than organically-managed and uncultivated soils (collectively). Mean and median WHC at both depths is also approximately 20-30% lower on conventional farms. Average SOC levels for organic and uncultivated soils are similar and significantly higher than conventional soils for the upper sample depth by at least double; in particular, organically-cultivated soils have 120% higher levels and uncultivated soils have 170% higher levels. Although no significant differences are observed at the lower depth due to outliers and high variation, average values for both of these groups are approximately double that of conventional soils.

However, while nitrogen concentration does not significantly differ between groups at the upper depth, higher median concentrations by at least 2x were seen on organic and uncultivated soils in comparison to conventionally-managed soils. Significant differences are observed at the lower depth, where median values for organic and uncultivated soils are 4-5x higher than seen for conventional soils. It can be seen that some uncultivated soil samples at the lower depth emerge as clear outliers with N concentrations (16.4-34.8 mg g^{-1}) much higher than all other soil samples, which had concentrations mostly <5 mg g^{-1} (Figure 70). These outlier data points were from different landscapes on different farms, sampled from a hedgerow, grassy field margin, and two woodlands. Outlier N values were also observed at the upper depth for all management categories, and again these were found on different farms and different landscapes.

Regarding C/N ratios, no significant differences are observed between management types. Median values are mostly similar between management types, although conventional farms have slightly higher C/N ratios at the upper depth (by 7%) and the lower depth (by 20%) than other groups. Median C/N ratios are similar between groups even though SOC concentrations are higher on uncultivated and organic soils due to the higher levels of N also seen on these soils. As also seen within urban and rural groupings (Figure 68), outliers with exceptionally high values exist throughout this dataset across uncultivated, organic, and conventional soils (Figure 70). Further statistical analysis was performed on C/N data excluding outliers (all values <1 and ≥60), and again, significant differences are not observed (Table 67; Appendix G, Figure 76).

Considering trace / heavy metal concentrations as provided in Figure 71, few significant differences between management types are identified. Organic soils have significantly higher Cd and Zn concentrations than uncultivated soils at the lower and upper depth, respectively. Cu and As concentrations are highly variable for conventional soils, with mean values anywhere from 35-75% higher than uncultivated and organic soils. However, median values are generally more similar. The exception is for Cu at the lower sampling depth, where conventionally-managed soils have significantly 37% higher concentrations than uncultivated soils. No significant differences between heavy metal concentrations on organic and conventionally managed soils are observed.

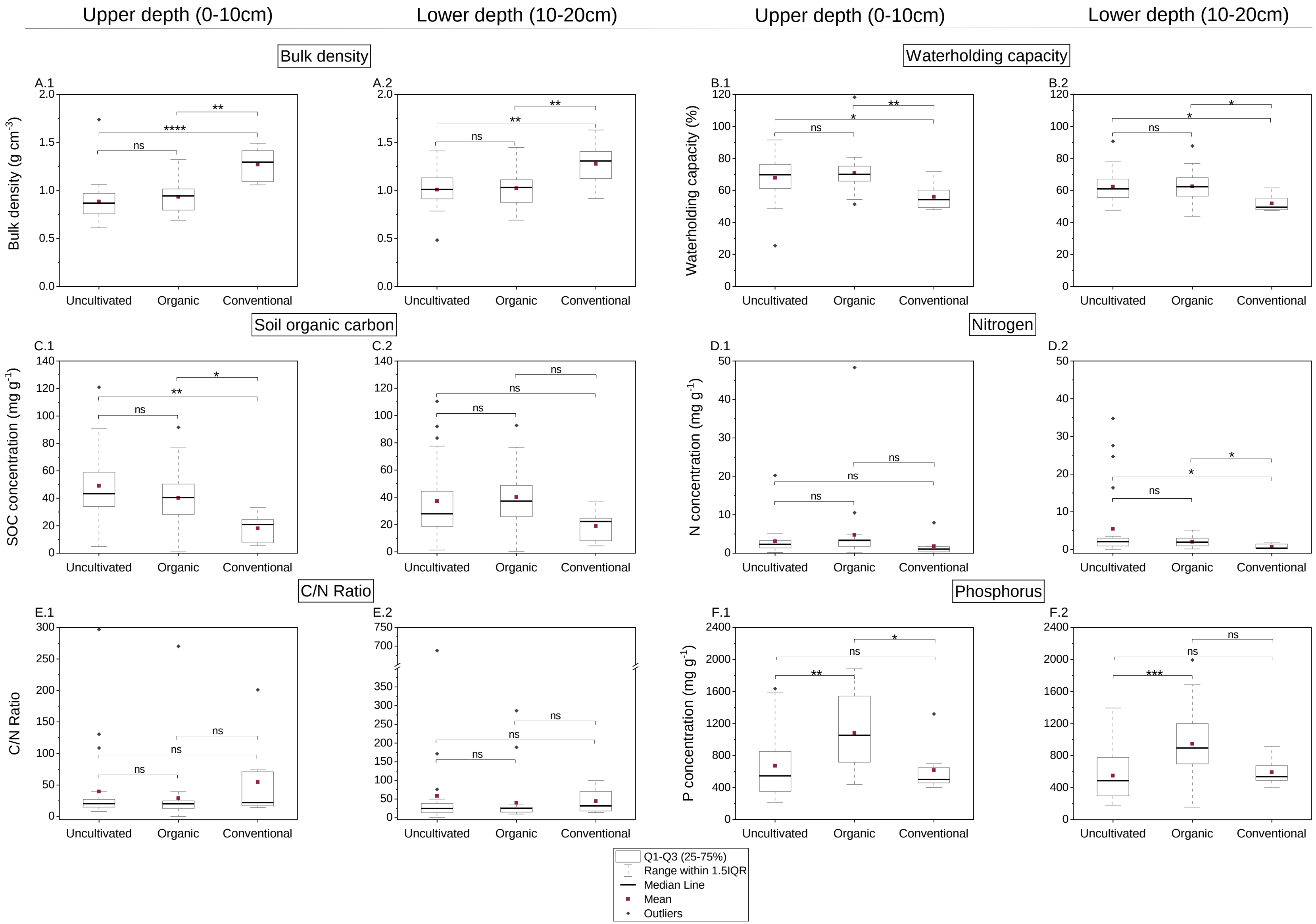

Figure 70 *- Box plots showing medians, means, interquartile ranges, and outliers for soil properties and major nutrients from uncultivated, organically-managed, and conventional soils. Subplots A-F show bulk density, water-holding capacity, soil organic carbon, N concentration, C/N ratio, and P concentration, respectively. Subplots numbered with a (1) refer to soils sampled at the upper depth (0-10 cm) and (2) refer to those sampled at the lower depth (10-20cm). Significant differences between groups are indicated by asterisks; no significant differences are denoted by 'ns.'*

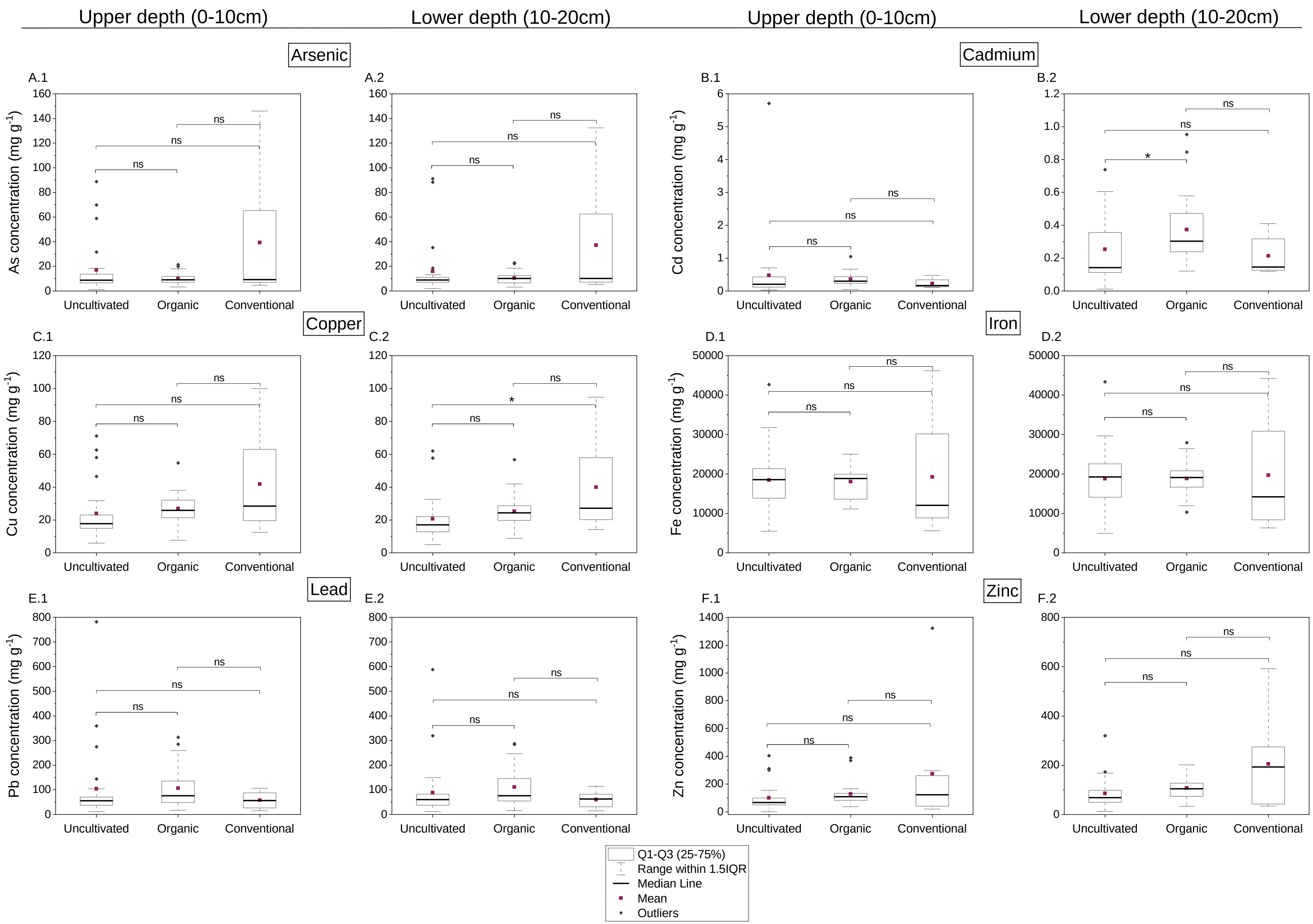

Figure 71 *- Box plots showing medians, means, interquartile ranges, and outliers for trace / heavy metal concentrations from uncultivated, organically-managed, and conventional soils. Subplots A-F show concentrations of As, Cd, Cu, Fe, Pb, and Zn, bulk density, respectively. Subplots numbered with a (1) refer to soils sampled at the upper depth (0-10 cm) and (2) refer to those sampled at the lower depth (10-20cm). Significant differences between groups are indicated by asterisks; no significant differences are denoted by 'ns.'*

4.1.3 Soil organic carbon (SOC) storage

Median SOC storage to a 10cm depth (1) and 20cm depth (2) are presented across different landscapes and management types Figure 72-A and Figure 72-B, respectively. When storage is considered for the 20cm depth rather than the 10cm depth, this generally results in at least double the SOC storage, as expected.

SOC storage levels are fairly similar between management types at the 10cm depth (Figure 72-A.1), ranging from 5.5-7.5 kg SOC m^{-2}. At the 20cm depth (Figure 72-A.2), more variation is seen although levels are still similar; the highest SOC storage is achieved in polytunnel soils (15.7 kg m^{-2}), then herbal leys (13.2 kg m^{-2}), orchards (13.1 kg m^{-2}), and hedgerows (12.6 kg m^{-2}). The lowest C storage (10.1 kg m^{-2}) is observed for the grassy landscapes (including field margins, pastures, and unmanaged fields). Interestingly, although hedgerows and woodlands showed the highest SOC storage to the 10cm depth, other cultivated land areas (polytunnels, leys, and orchards in particular) had higher SOC storage when considering the larger depth.

When comparing based on management type, different trends are again observed for the two different depths. Uncultivated and organically-cultivated land areas have similar SOC storage levels to the 10cm depth (approximately 6.8 kg SOC m^{-2}), which are nearly 40% higher than that seen for conventionally-managed land at 4.9 kg m^{-2} (Figure 72-B.1). On the other hand, when considering storage to the 20cm depth, organically-cultivated land clearly results in the highest SOC storage at 13.6 kg SOC m^{-2}, compared to levels of approximately 10.4 kg m^{-2} for uncultivated and conventional land (Figure 72-B.2). This higher storage on organic land is likely driven in part by the higher SOC storage contributed from polytunnel and orchard soils, as these landscapes were only present on organic farms.

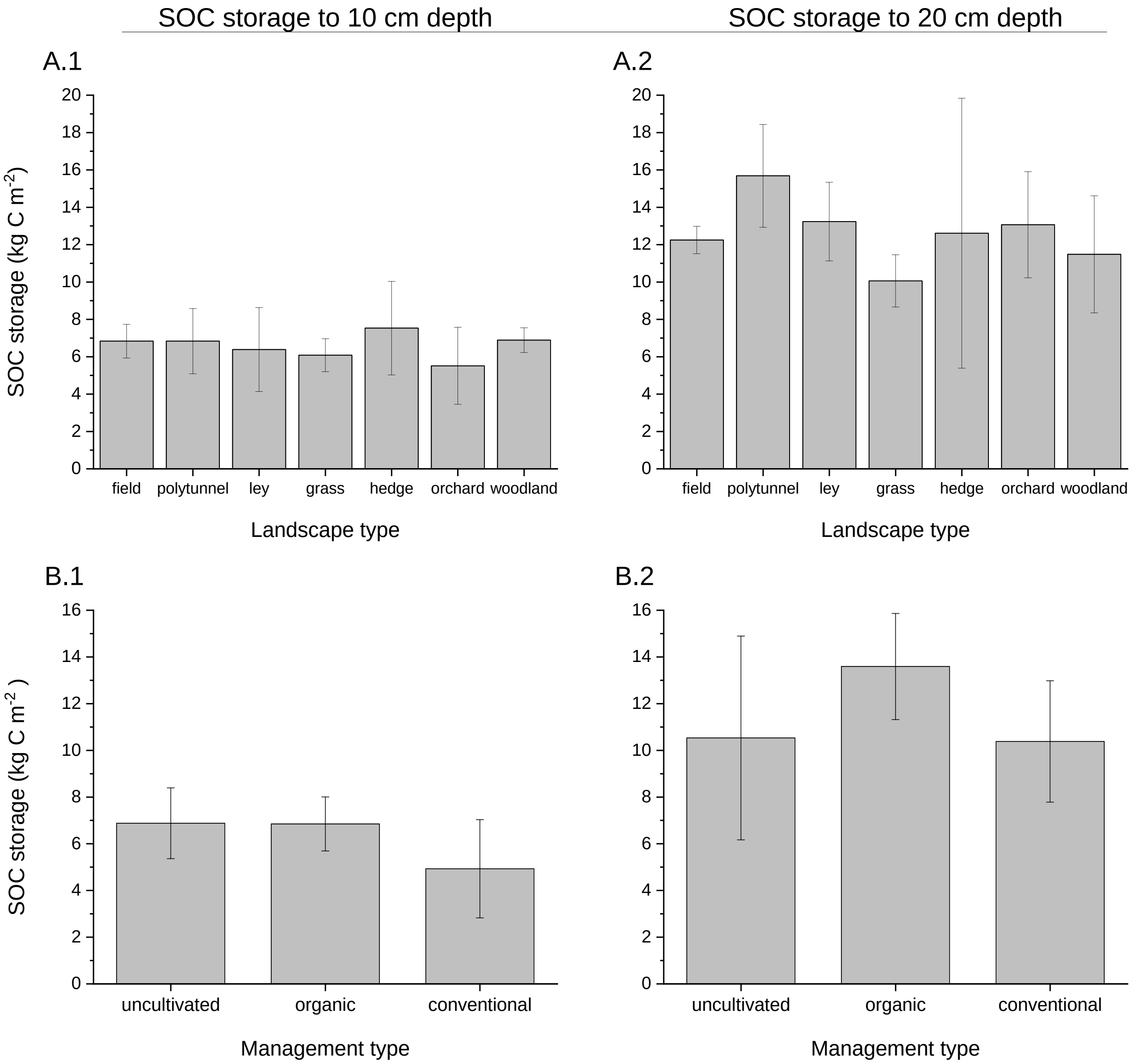

Figure 72 *- Median soil organic carbon (SOC) storage (kg m^{-2}) to (1) 10 cm depth and (2) 20 cm depth, per A. landscape type and B. management type. Error bars indicate interquartile range.*

4.2 Discussion

This study evaluated key soil health metrics and heavy metal concentrations across ten UK farms, characterised based on urbanisation (urbanised vs. rural) and management type (conventional, organic, or uncultivated). The main soil health metrics varied most when comparing farms based on management type, whilst heavy metal concentrations showed significant differences based mainly on urbanisation.

In relation to the past study evaluating the sustainability of these farms through lifecycle assessments (Chapter 3), this research highlights key ecosystem services and disservices not captured within LCA frameworks. In particular, organically-managed and uncultivated land areas on farms tended to have better soil health and thus likely contribute more to supporting ecosystem services than conventionally-managed land. This is a key insight in light of many LCA studies that have highlighted higher environmental impacts from organic farms vs. conventional farms when evaluated on a product-basis (Tuomisto *et al.*, 2012; Meier *et al.*, 2015). Although certain UK organic farming groups (mainly rural) did have lower environmental impacts than conventional farms within this research (Chapter 3), it is still important to outline that there are obvious benefits from organic farming not being captured in LCA frameworks. Equally important to note are the benefits that come from the non-productive areas on farms, such as woodlands, hedgerows, long-term herbal leys, and field margins. As LCAs tend to focus on the lifecycle of one crop, as necessary for comparative purposes, this means the value of uncultivated spaces on farms are generally not attributed, even though these may be essential parts of land management plans on both organic and conventional farms.

However, the higher heavy metal concentrations seen on urban farms also highlights that there may be important ecosystem *dis*services not captured in LCA frameworks. Although LCAs take into account heavy metal concentrations from soil inputs, which relate mainly to human toxicity and ecotoxicity impacts, the potential for growing crops in previously contaminated soil is not realised. The finding of relatively high heavy metal concentrations in urban soils is not new, and corresponds with a wealth of research for UK cities (Culbard *et al.*, 1988; Alloway, 2004; Ander *et al.*, 2013; Crispo *et al.*, 2021) and other cities across the globe (Antisari *et al.*, 2015; Kosheleva and Nikiforova, 2016). The propensity of these heavy metals to contribute to environmental and health-related harms, however, is not clear from this study; still, high concentrations do show a need for further investigation.

The main findings of this research are explored in more detail in the following sections. Comparisons are made to national values for soil health parameters and heavy metals as found in the latest Countryside Survey in 2007, which includes measurements for soil quality indicators across Great Britain (Emmett *et al.*, 2010). This survey measures soil parameters to a 15 cm depth; thus, this will be compared with values found for the upper soil depth (0-10cm) in this study. Additionally, this discussion provides examples from specific farm context, supported by Appendix H (Table 82 and Table 83), which includes metrics for undisturbed and cultivated soils on each individual farm.

4.2.1 Main soil health metrics

The key soil health metrics evaluated in this study include: bulk density, water-holding capacity (WHC), soil organic carbon (SOC), total nitrogen, C/N ratios, and total phosphorous. These metrics are used to provide indication of important ecosystem services such as flood mitigation, soil structure, erosion control, carbon storage, and soil fertility, as well as indication of any signs of soil degradation as a result of agricultural activity. The main differences in these metrics are observed between management types, and thus the following sections focus mainly on comparisons between uncultivated, organic, and conventional soils.

4.2.1.1 Bulk density and water-holding capacity

Organically-managed and uncultivated soils had significantly lower average bulk densities and higher water-holding capacities by 20-30% compared to conventional soils, indicating that these soils are contributing more positively to ecosystem services such as flood mitigation, improved soil structure, and erosion control. The observed differences are likely associated with reduced soil disturbance and higher levels of organic matter for uncultivated and organic soils. This is supported by the fact that SOC is often found to be negatively correlated with bulk density and positively correlated with WHC (Emmett *et al.*, 2010; Libohova *et al.*, 2018), which was also seen in this study through PCA results (Figure 66-A). Additionally, bulk densities were generally lower, whilst WHC and SOC were generally higher, at the upper sampling depth vs. the lower depth, indicating that recently-incorporated organic matter in the topsoil is likely driving these improved soil qualities (Leifeld and Fuhrer, 2010). Urban soils also had slightly lower bulk densities than rural soils, although only significant at the lower depth; this difference is likely related more to higher organic matter additions on these farms rather than urbanisation, especially since urban soils often show signs of compaction (Beniston and Lal, 2012; Kumar and Hundal, 2016). This suggests that organic farm management on these urban soils actually may be contributing to improved soil structure.

Compared to national values, average bulk density (0-10cm) from conventionally-managed soils (1.27 g cm^{-3}) are similar to the average values for arable and horticultural farmland across Great Britain (1.23 g cm^{-3}) (Emmett *et al.*, 2010). However, mean values for organically-managed (0.93 g cm^{-3}) and uncultivated land areas (0.89 g cm^{-3}) are more similar to values for neutral grassland areas (0.90 g cm^{-3}) (Emmett *et al.*, 2010), as well as those reported from allotments across the UK (0.92 g cm^{-3}) (Dobson *et al.*, 2021).

The higher bulk densities and lower WHCs observed on conventional soils, versus organic and undisturbed soils, may be indicative of increased soil compaction and thus potential soil degradation (Edmondson *et al.*, 2011). This is likely a result of more frequent use of heavier machinery on the farm (Shah *et al.*, 2017). Indeed, the organic farms included in this study are mostly small-scale (Table 59) and use mainly two-wheel tractors or very small tractors (<50 HP), if using machinery at all.

Compaction can impact crop growth by impeding root penetration in the soil. Crop growth may begin to be negatively impacted around critical bulk density values of 1.40-1.75 g cm^{-3}, depending on soil type (Daddow and Warrington, 1983). In this study, two conventional farms were seen to have bulk density values near the minimum of this range (R-C-3 and R-C-5); thus, compaction may be an important concern for these farms and an indication of long-term soil degradation. The fact that the bulk densities (at the upper level) for these farms are 40-50% higher than other undisturbed areas on the farm further suggests degradation of these soils (Appendix H, Table 82). It is interesting to point out that higher bulk density values for

R-C-5 were actually seen on their organic fields. However, it should be noted that this field was only in its first year of organic certification, after being in two years of herbal leys for the required organic transition period. Additionally, this field actually sees higher tractor activity than conventional fields due to the need for mechanical weeding, as confirmed by the farmer. This likely influences the higher compaction on this field, which is not seen on the other smaller-scale organic farms that use less machinery and have also been managed under organic principles for anywhere from 10-25 years.

Besides affecting crop growth, compaction is a concern because it can lead to increased runoff, flooding, and erosion by limiting water infiltration (Shah *et al.*, 2017). This is supported by the negative correlation between WHC and bulk density seen in the PCA results (Figure 66-A) and highlighted by previous research (Hamza and Anderson, 2005; Libohova *et al.*, 2018). The farms with the highest bulk densities (R-C-3 and R-C-5) are located in nitrate-vulnerable zones as defined by the UK Environment Agency, with specific concerns for leaching to surface water (UK Environment Agency, 2021). Therefore, improving soil water storage on these farms is critical to prevent environmentally-damaging nutrient losses.

Compacted soils are also more likely to be water-logged, which can create anaerobic conditions that support the loss of N as N_2O or N_2 through denitrification (Goulding, Jarvis and Whitmore, 2008), thus resulting in potential greenhouse gas emissions from these soils. As R-C-3 and R-C-5 also have soil types more prone to water-logging due to low permeability and high groundwater tables, respectively, (based on soilscapes: Cranfield University, 2011), the high bulk densities seen on these farms are even more concerning. It is therefore clear that the effective timing of fertiliser applications at the right amount is critical on these farms (Johnston and Bruulsema, 2014). Additionally, the relatively high WHC and low bulk density seen for uncultivated land areas overall (Figure 70) shows that these spaces may be critical for natural flood mitigation across all farms; hedgerows or strips of woodland in particular are useful to intercept runoff and also provide a barrier to wind and rain, thus limiting erosion (UK Environment Agency, 2019).

4.2.1.2 Soil organic carbon (SOC)

Higher levels of SOC (or SOM) are almost always desirable, because SOC is beneficial for a wide range of physical, chemical, and soil biological properties, supports crop growth, and serves as the basis for most soil-based ecosystem services (Powlson *et al.*, 2011; Lal, 2014). This study shows that organically-managed and uncultivated soils have double to triple the SOC concentration of conventionally-managed soils at both sampling depths, although higher (and significant) at the upper depth. This finding is consistent with previous reviews assessing organic and conventional farming systems, which have also found higher levels of SOC or SOM on organic farms (Leifeld and Fuhrer, 2010; Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012). This study also found 27% higher SOC concentrations on urbanised vs. rural soils at the upper depth, although related primarily to management decisions rather than urbanisation. Indeed, all urban farms were also organic farms, hence why the difference between these systems is somewhat diluted. Overall, these results indicate that uncultivated or organically-cultivated spaces on farms are contributing more to the many ecosystem services supported by SOC, such as maintaining soil structure, reducing erosion, resisting compaction, increasing water retention and availability, and thus aiding in flood mitigation, supporting soil biodiversity, purifying water, and immobilising or denaturing pollutants (Lal, 2014). Higher SOC improves the ability of soil to react and adapt to environmental stresses (‘soil resiliency’), which is crucial in the face of climate change (Lal, 2014).

The higher SOC concentrations on urban farms and in organic and uncultivated land areas is likely attributed to the higher organic matter additions in these systems, as OM additions and SOC are associated in a positive linear relationship (Johnston, 1975; Johnston, Poulton and Coleman, 2009). This is further supported by past reviews highlighting higher OM inputs on organic farms as the main driver of higher SOC concentrations in comparison to conventional farming systems, which usually have lower OM inputs (Leifeld and Fuhrer, 2010; Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012). Previous studies have found that changes in SOC as a result of converting to organic agriculture occur mainly in the top 15-20cm, with changes in SOC at lower soil depths generally not significant even over long-term experiments (10-20+ years) (Leifeld and Fuhrer, 2010; Blanco-Canqui, Francis and Galusha, 2017). This is also reflected in this study as differences in SOC concentrations between management and urbanisation types are only significant at the upper depth (0-10cm), therefore likely a consequence of topsoil OM additions.

OM additions on organic farms in this study come from consistent and relatively high applications of organic fertilisers like compost and manure, as well as from incorporation of cover crop residues and organic mulches. The same is true for urban farms, with findings from the LCA study (Chapter 3) showing that urban and peri-urban organic farms tend to apply higher levels of compost and manure than rural organic farms, thus potentially driving SOC even higher. The high SOC concentrations on urban farms also relates well with previous UK studies finding higher SOC in urban gardens and allotments than on conventional, rural farms due to higher OM additions (Edmondson *et al.*, 2014; Dobson *et al.*, 2021). For uncultivated land areas, higher SOC is likely associated with less soil disturbance and, for hedgerows and woodlands, OM additions from leaf litter (Edmondson *et al.*, 2014).

Differences in SOC storage levels (kg m^{-2} to 20cm depth) between management types are less severe than seen for concentrations. Numerically, organically-cultivated soils have the highest SOC storage levels to 20cm, approximately 30% more than conventional and uncultivated soils, which are similar. The comparison with the latter case is particularly interesting as normally one would expect uncultivated land areas to have the highest SOC storage (Emmett *et al.*, 2010); this highlights that organic farming areas may be crucial stores of soil C that are not specifically assessed in national surveys (Emmett *et al.*, 2010). Regarding SOC storage values (to 20 cm depth) across landscapes, it is interesting to see that cultivated fields store largely similar amounts of carbon (12.2 kg m^{-2}) as orchards (13.1 kg m^{-2}), hedgerows (12.6 kg m^{-2}), and woodlands (11.5 kg m^{-2}). The highest storage is seen for soils under polytunnels (15.7 kg m^{-2}), found on organic farms only, which is likely due to the large amounts of compost used to make raised beds, low machinery use (and thus lower soil disturbance), and increased protection from wind and rain.

The fact that relatively large differences in SOC concentrations, but relatively small differences in SOC densities (i.e., SOC storage), are observed between organic and conventional soils is related to differences in bulk densities, as also highlighted in past studies (Leifeld and Fuhrer, 2010). As conventional soils have higher bulk densities, this means that there is a larger soil mass contributing to SOC per any given unit of volume, which can outweigh the lower concentrations of SOC per unit weight of soil. However, as higher bulk densities are undesirable, higher SOC densities do not necessarily point to better soil health, although they are useful to examine for carbon accounting purposes. Lower SOC storage in some uncultivated soils may also be attributed to higher levels of debris or stones (often removed in farmed soils over time), which are counted for within the volume measurements but do not contribute to C storage.

Compared to national values, SOC concentrations for conventional soils (18 g kg^{-1} at the upper depth) are almost half of that observed for arable and horticultural soils in England (30 g kg^{-1}) (Emmett *et al.*, 2010), although more similar to the 23.5 g kg^{-1} median value for UK arable and horticultural soils found by Dobson et al. (2021). Alternatively, when approximating SOC storage values to the 0-15 cm depth to compare with the Countryside Survey, the mean and median SOC storage from conventional farms (6.5 and 7.6 kg m^{-2}, respectively) is higher than other arable and horticultural farms nationally (4.7 kg m^{-2}) and more comparable to values for fertile and infertile grasslands (6.1-6.9 kg m^{-2}, respectively); this again can be attributed to the higher bulk densities seen on these farms.

SOC concentrations on organically-managed and uncultivated soils (40 and 49 g kg^{-1}, respectively) are substantially higher than that seen for arable and horticultural soils nationally, being more similar to values found for fertile grasslands in England (45 g kg^{-1}) (Emmett *et al.*, 2010). For SOC storage (as approximated to the 15 cm depth), mean and median levels from uncultivated and organic soils are similar (9-11 kg m^{-2}) and are markedly higher than most English habitats examined in the Countryside Survey, even generating values above that of woodlands (7-8 kg m^{-2}) and being more comparable to heathland and bogland (9.1 kg m^{-2}).

SOC is usually seen as a proxy measurement for ecosystem services related to carbon sequestration (Lal, 2004; Leifeld and Fuhrer, 2010; Gattinger *et al.*, 2012). However, care should be taken when making this connection, especially with agricultural soils. For an increase in SOC to contribute to climate change mitigation, this must result in a true net transfer of C from the atmosphere to the soil (i.e., sequestration), such as through slower decomposition of SOM or additional photosynthesis (Powlson *et al.*, 2011; Stockmann *et al.*, 2013). In many cases, increases in SOC simply reflect a transfer of carbon from one terrestrial C pool, or one location, to another (e.g., spreading of manure or compost produced on other sites) (Powlson *et al.*, 2011). SOC in the first 30 cm is most susceptible to change as a result of management practices, but this topsoil SOC is also readily decomposable and thus generally considered to be in the labile (or ‘fast’) carbon pool, with decomposition occurring in the timescale of a few days to a few years (Stockmann *et al.*, 2013; Clara *et al.*, 2017). Increased SOC as a result of increased OM additions, such as seen for organically-cultivated soils in this study, is therefore likely to be decomposed relatively quickly and returned back to the atmosphere as CO_2. Thus, the SOC concentrations measured in this study (to the 20cm depth) do not necessarily reflect long-term soil carbon change.

However, because the conversion of natural ecosystems to agricultural land is associated with stark decreases in SOC (40-60%), particularly in the top 60 cm (Guo and Gifford, 2002; Lal, 2004; Stockmann *et al.*, 2013), there is still an obvious potential to replenish SOC stocks in potentially depleted agricultural soils (Dungait *et al.*, 2012). Approximately half of the SOC in England is stored within the top 30cm (Gregory *et al.*, 2011), indicating that agricultural topsoils are a key area of opportunity for carbon storage but also a risk, as this SOC can easily be lost from soil degradation processes (Powlson *et al.*, 2011). Consistent application of large amounts of organic matter can shift equilibrium levels of SOC within the upper layers of the soil, but this happens over a long time-frame and can take more than 100 years (Johnston, Poulton and Coleman, 2009).

Even if the higher SOC concentrations or densities seen on organic farms do not result in higher long-term C storage, an important point to note is that the cultivated soils on these farms have mostly maintained SOC concentrations in line with respective soils in undisturbed landscapes (Appendix H, Table 82). Indeed, across all organic farms in this study (excluding R-C-5's organic land area), SOC concentrations and storage levels on cultivated fields and polytunnels are usually not lower than undisturbed land areas by more than 12% at both depths; in many cases, SOC levels in cultivated soils are actually higher than in undisturbed soils. The exception is for PU-O-1 and R-O-2, which respectively see approximately 30% and 60% lower SOC concentrations / storage values on cultivated areas at the upper depth; however, levels are still relatively high and in the range of other organic farms.

On the other hand, cultivated soils on conventional farms are seeing depletion of SOC concentrations (0-10cm) by 60-80% compared with undistributed soils (Appendix H, Table 82); this was also observed on R-C-5's organic field, which likely was not under organic management long enough to see SOC changes. Differences are similar at the lower depth, except for R-C-3, where SOC levels are maintained. Decreases in SOC storage (to 20 cm) on these farms are between 20-70%. The lowest SOC concentrations are observed again on R-C-3 and R-C-5 (organic and conventional), with values an order of magnitude lower than all other farms. At the upper depth, SOC concentrations for these farms (6-8 g kg^{-1}) are less than the critical level of 10-15 g kg^{-1} defined by Lal (2015) as necessary to reduce soil degradation risks. R-C-5 shows further depletion of SOC at the lower soil depth (2-5 g kg^{-1}), which is particularly concerning as lower soil depths are generally expected to contribute more to stable C pools (Stockmann *et al.*, 2013). This again shows that the agricultural land on these farms, which also had some of the highest bulk densities and lowest WHCs, may be at risk of soil degradation.

Of course, there are other factors driving SOC concentrations that should be considered. In particular, SOC (and thus bulk density and WHC) is related to soil texture, with sandy soils reaching lower equilibrium SOC states than clayey soils (Johnston, Poulton and Coleman, 2009). R-C-3 has a high proportion of black sand in their field soil, and thus this could be a potential driver of lower SOC; however, differences of 80% SOC (at 0-10 cm) are observed between cultivated and undisturbed soils in the same field, which suggest likely depletion of OM due to agricultural activity (Appendix H, Table 82). Additionally, the fact that this is not observed at the lower sampling depth, and that this field was sampled later in the year (after harvest), suggests that a replenishment of OM through cover cropping or other fertilisers could help remediate this soil.

Overall, these findings show that SOC concentrations have either improved or reduced only slightly in cultivated soils on organic farms (vs. undisturbed counterparts), which indicates that these farming systems may be preventing the loss of SOC and the related soil degradation usually associated with agricultural land transformations (Guo and Gifford, 2002; Emmett *et al.*, 2010). This is mostly related to high OM additions and minimal soil disturbance on these farms over the long-term, which is why high SOC was not seen on the R-C-5's large-scale organic field, being organic only for a short time (2 years of conversion) and relying more on tractor-based mechanical weeding than the other small-scale organic farms. Additionally, R-C-4 showed SOC concentration and storage levels similar to organic farms, which may be a result of steps the farmer has taken to increase organic matter additions through cover cropping and reduce soil disturbance through minimum tillage efforts. Similar management changes made on other conventional farms may also help to rebuild SOC and soil health (Powlson *et al.*, 2011).

4.2.1.3 Total nitrogen

Total N did not differ significantly between urbanisation or management types in most cases. Numerically, conventional soils had lower median total N concentrations than uncultivated and organic soils by approximately 50-70% at the upper depth and 80% at the lower depth, with significant differences observed in the latter case when compared to uncultivated soils. Median total N for conventional soils (1.09 mg g^{-1} at the upper depth) is also approximately half of that seen for arable and horticultural soils nationally (2.5 mg g^{-1}) (Emmett *et al.*, 2010); total N at the lower depth on these farms is even less (0.34 mg g^{-1}). As SOM is the largest reservoir of N in the soil (Goulding, Jarvis and Whitmore, 2008), low levels of total N are likely related to the lower levels of SOM in conventionally-cultivated soils, as SOC concentrations for these soils are also about half of those observed nationally (Emmett *et al.*, 2010). Although higher SOC concentrations would be desirable on these farms, the fact that total N was relatively low when sampled at the end of the season reflects an optimal application of fertiliser N to match crop needs, highlighting the role that these farms are playing in reducing potential leaching risks.

On the other hand, N concentrations for organic and uncultivated soils (1.9-3.3 mg g^{-1} at both depths) are more similar to values for agricultural soils nationally, but lower than values seen for grasslands or woodlands (3.9-6.6 mg g^{-1}) (Emmett *et al.*, 2010). Median values for urban farms are slightly higher (approximately 4.0 mg g^{-1} at both depths) and are similar to those seen for UK urban allotments (3.7 mg g^{-1}) (Dobson *et al.*, 2021). Total N on urban farms is also approximately 35-50% higher than on rural farms (depending on depth), but this difference is not significant and is likely related more to management practices (higher OM additions on urban farms) than urban soil properties. Indeed, although the N input on organic farms is usually assumed to be lower than on conventional farms (Kirchmann and Bergström, 2001), the previous lifecycle inventories for the organic farms in this study showed similar or even higher total N inputs in some cases, mainly from organic materials such composts, manures, and cover crop residues (see: Section 2.1.7.3.2 and 2.1.7.4.2) . As inputs of soluble N were usually much lower, this indicates that most N in these organically-farmed soils is likely held in the organic matter, and depending on mineralisation rates, N availability for crops may still be an issue (Watson, Atkinson, *et al.*, 2002).

The high levels of total N often observed on soils that see large applications of composts and other organic inputs, such as urban farms and gardens, has been highlighted as a potential risk for nutrient losses to the environment where this exceeds crop needs (Small *et al.*, 2019; Wielemaker *et al.*, 2019). N applied in excess of crop demand is either retained in the soil in organic matter, lost to the atmosphere through volatilisation (as ammonia) or denitrification (as N_2O or N_2), or lost to water via nitrate leaching (Goulding, Jarvis and Whitmore, 2008).

Thus, over-application of N is as much of a concern as depleted N, particularly since eight out of ten farms in the study exist in nitrate-vulnerable zones, and seven have soil types prone to water-logging and thus loss of N via denitrification.

However, the higher levels of total N seen on organic and urban farms do not necessarily mean that these soils are more likely to contribute to leaching, since much of this N is likely contained in the organic matter and made available to crops over time via mineralisation (Stockdale *et al.*, 2002). The Countryside Survey in particular highlighted that total N is not an appropriate measure to estimate eutrophication or leaching losses; indeed, although the survey found that total N concentrations for arable and horticultural soils were relatively low, this land class actually had higher nitrate concentrations than several other habitats , which is more relevant to leaching risk (Emmett *et al.*, 2010). However, organic N can also be rapidly mineralised at certain times of the year and thus contribute to leaching risk (Kirchmann and Bergström, 2001); for example, it has been found that up to 50% of N in leguminous residues can be mineralised in the first two months after incorporation (Kirchmann and Bergqvist, 1989). Thus, matching times of high N mineralisation with nitrogen-demanding crops in the rotation is critical and seen as one of the main challenges for organic farms (Watson, Atkinson, *et al.*, 2002). In future studies, measures of mineralisable and nitrate N would be useful to provide further insight to potential nutrient losses in organic and conventional systems.

4.2.1.4 C/N ratios

In this study, median C/N ratios are similar between management types (20-22 at the upper depth), but are much higher than that previously seen for arable and horticultural land (11), grasslands (12-13), and even broadleaf woodlands (16) across Great Britain (Emmett *et al.*, 2010). Similar median values of 18-19 across management types are seen even when excluding outlier values (Appendix G, Figure 76). These values are more similar to the median C/N ratio of 16 found in UK urban allotments (Dobson *et al.*, 2021) and also close to the C/N ratio of 17 found for temperate croplands internationally (Amorim *et al.*, 2022). The relatively high C/N ratios are driven mainly by the relatively low N concentrations seen across many farms, as well as relatively high SOC concentrations for organic and uncultivated soils. Urban soils also see significantly higher C/N ratios (21) compared to rural soils (14) at the upper depth, likely attributed to higher SOC concentrations as a result of larger organic fertiliser applications than seen on rural farms (see: Section 2.1.7.3.2 and 2.1.7.4.2).

Several outlier C/N ratios existed in this study, mainly due to exceptionally low N values of 0.10-0.20 mg g^{-1} observed on certain farms. These low N values resulted in C/N ratios in the range of 60-688, which at the upper end are more similar to that seen for charcoal or biochar than natural soils (Bonanomi *et al.*, 2017). This leads the author to believe that some samples may have been rich in black carbon or included portions of charcoal or other burnt residue. This is supported by the fact that on one farm with high C/N values (R-C-3), the farmer highlighted that cinder is commonly found in the soil because the land was historically spread with sewage and industrial waste from nearby industrial cities. The other farm that saw high C/N values across most landscapes (R-O-2) is located in an area where arable farming is historically prolific; stubble burning was common practice on UK arable farms in the late 1900s until this was legally prohibited in 1993 (Bullen, 1974; UK Government, 1993). Thus, this historic management practice may have also left charcoal residues in many UK agricultural soils.

C/N ratios are used as a basic indicator of SOM decomposability and nutrient cycling (Amorim *et al.*, 2022). If C/N ratios are too high, this can limit plant N availability due to microbial immobilisation; if too low, N will be rapidly mineralised from the soil, which can result in nutrient losses if this is not taken up by crops (Dungait *et al.*, 2012). Thus, there are ideal values at which nutrient cycling in the soil is optimised, which is often cited as at least 10 (USDA NRCS, 2022), although these are context-specific and differ depending on soil pH, soil texture, and climatic factors (Lucas and Davis, 1961; Haney *et al.*, 2012; Amorim *et al.*, 2022). Clayey and basic soils generally exhibit lower C/N ratios, whilst sandy and acidic soils see higher C/N ratios (Amorim *et al.*, 2022). There are also critical maximum C/N ratios to consider, above which N immobilisation is likely to occur (Haney *et al.*, 2012). These critical values differ, but are often cited in the range of 25-30 (USDA NRSC, 2011; Haney *et al.*, 2012). Excluding the farms with major outlier samples (R-O-2, R-C-3, and R-C-4), nearly all other farms have C/N ratios in the range of 10-25 (Appendix H, Table 82); however, some have values near the extremes, indicating potential risk of excessive mineralisation (and thus potential leaching) or immobilisation (and thus potential N deficiency). The latter case is particularly a concern for crop production on organic farms, which is usually limited by plant available N (Seufert, Ramankutty and Foley, 2012).

4.2.1.5 Phosphorus

Organically-managed soils show significantly higher total P concentrations by nearly double at both depths, in comparison to conventionally-managed and uncultivated soils; no significant differences are seen between urbanised and rural soils. This shows that total P concentrations are influenced mainly by management decisions. The higher total P concentrations seen on organically-farmed soils actually contests many prior studies, where total and available/extractable P concentrations are often seen to be lower on organic vs. conventional farming systems (Maeder *et al.*, 2002; Gosling and Shepherd, 2005) or decrease over time with organic management (Goulding, Stockdale and Watson, 2008; Keller *et al.*, 2012). This has led to the notion that organic farming may be 'mining' available soil P reserves and actually depleting soil fertility over time (Goulding, Stockdale and Watson, 2008; Cooper *et al.*, 2018), which is a concern because the majority of natural soil P is inaccessible to plants (Sharpley, 2000; Darch *et al.*, 2014).

The relatively low soil P seen in organic farming systems is usually related to negative P nutrient budgets, where more P is exported in crop biomass than is added via fertilisers (Oehl *et al.*, 2002; Watson, Bengtsson, *et al.*, 2002; Keller *et al.*, 2012). However, this is likely more relevant to arable or livestock farms than those in this study. Indeed, Watson et al. (2002b)'s review of organic farm nutrient budgets actually showed large P surpluses in many horticultural systems due to the use of imported manure. Similarly, recent studies of urban gardens have found high P surpluses from the application of large amounts composts and other organic amendments (Small *et al.*, 2019; Wielemaker *et al.*, 2019).

The organic farms in this study are also characterised by high levels of manure and compost application, as elucidated through the lifecycle inventories, and this likely explains the high total P concentrations in the cultivated soils on these farms. Because organic inputs on these farms are likely applied to satisfy crop N demand, this can result in excess P in the system

(Kleinman *et al.*, 2011; Lemming *et al.*, 2019; Small *et al.*, 2019). Compost use in particular may lead to P accumulation in the topsoil due to low soil mobility (i.e., slow progression through soil depth) (Lemming *et al.*, 2019; Hu *et al.*, 2022). For example, Hu et al. (2022) found higher levels of total P in soils fertilised with compost compared to mineral P fertilisers after 20 years of treatment.

A particular concern for organic farming systems is the build-up of nutrients in organic or otherwise unavailable forms whilst depleting plant-available nutrient pools (Stockdale *et al.*, 2002). Indeed, the relatively high total N concentrations seen on organically-managed soils (as compared to conventional) is likely indicative of build-up within SOM. As for total N, however, the measure of total P does not provide direct insight to plant-available P, which is better observed through measures of extractable P (Darch *et al.*, 2014); thus, depletion of plant-available P may still a concern for these organic farms despite high levels of total P. Studies examining the application of waste inputs (such as manure and compost) on soil P over time have shown that total P and available P are positively correlated, although fractions vary depending on the input (Lemming *et al.*, 2019). However, the amount of plant-available P provided by organic inputs is context-specific, and depends upon the specific fertiliser, as well as soil pH, texture, and other climatic factors (Sharpley, 2000; Lemming *et al.*, 2019). Indeed, some studies have found that compost-treated soils have low available P (Lemming *et al.*, 2019), whilst others have concluded that P from compost is as readily available as from other mineral fertilisers (Hu *et al.*, 2022).

An equally valid concern for the organic farms in this study is potential P nutrient losses to the environment due to consistent over-application of P and relatively high soil P concentrations (Kleinman *et al.*, 2011). P losses most commonly occur as dissolved or particulate P in run-off during storm events, particularly after a recent fertiliser application, thus affecting surface waterbodies; however, P leaching to groundwater can also occur (Fortune *et al.*, 2005; Withers *et al.*, 2009). Eutrophication risks from P are primarily associated with its water-extractable or soluble forms (Sharpley and Moyer, 2000). High risk would thus be expected following the application of water-soluble mineral fertilisers, although P losses from manures can be similar or even greater (Sharpley and Moyer, 2000; Kleinman *et al.*, 2002). On the other hand, compost is likely to pose a lower risk, although proper timing of nutrient application is always essential (Lemming *et al.*, 2019). Risk of nutrient loss can be further minimised through erosion control measures – for example, by maintaining soil cover and landscape features that can protect against wind and water, such as trees and hedgerows (Blanco-Canqui *et al.*, 2015; UK Environment Agency, 2019). In the future, evaluating extractable P fractions would be useful to indicate any risk of soil P depletion as well as nutrient losses (Sharpley and Moyer, 2000).

4.2.2 Soil metal concentrations

Soils act as a long-term sink for heavy metals. Unlike organic pollutants, heavy metals generally do not undergo any chemical or microbial degradation, and they also tend to have relatively low mobility (movement through soil depth) and bioavailability (Bolan and Duraisamy, 2003). Thus, heavy metals accumulate in soils over time. The most important natural process contributing to soil heavy metal concentrations is the weathering of bedrock, or soil parent material, which can result in elevated heavy metal concentrations in soils (Nowack *et al.*, 2001; Ander *et al.*, 2013). Thus, different soil parent materials on farms may contribute to different ‘baseline’ heavy metal concentrations (Ander *et al.*, 2013). However, soil heavy metal concentrations are also influenced by current and historic anthropogenic activities, including both industry and agriculture (Zwolak *et al.*, 2019).

This study found that urbanised soils had significantly higher concentrations of most heavy metals than rural soils. Past UK studies have also found relatively high concentrations of potentially harmful heavy metals in other types of urban agriculture sites, such as allotments and gardens, in some cases exceeding UK Government screening levels for allotment soils (Alloway, 2004; Giusti, 2011; Entwistle *et al.*, 2019; Crispo *et al.*, 2021). The largest differences between the metals in urban and rural soils in this study are seen for Pb, Zn, and Cd (in descending order), where urban soils have anywhere from 60-160% higher median concentrations than rural soils at both depths. Prior has research also highlighted these three heavy metals as common contaminants in urban soils (Culbard *et al.*, 1988; Sterrett *et al.*, 1996; Ander *et al.*, 2013).

On the other hand, few significant differences are observed between management types, indicating that elevated heavy metal concentrations on urban farms are likely more of a result of other anthropogenic activities in the area (e.g., industry) than agricultural practices. This is further supported by the fact that heavy metal concentrations in both the undisturbed areas and the cultivated areas on urban farms are higher than in most rural soils (Appendix H, Table 83). Additionally, the farms that are closest to urban centres (U-O-2, PU-O-1, and PU-O-2) generally see the highest heavy metal concentrations.

A major source of soil heavy metals like Pb, Zn, and Cd is atmospheric deposition from emissions of dust and aerosols generated by industrial activities, such as mining, smelting, and power generation; additional contributions come from the transport sector, industrial wastes and related surface runoff, and tyre wear on roads (Alloway, 2013; Zwolak *et al.*, 2019). These sources are mainly found to be more of a contributor in urban soils, as major current or post-industrial areas (Culbard *et al.*, 1988; Sterrett *et al.*, 1996; Zwolak *et al.*, 2019). Thus, these types of industrial heavy metal sources are likely the main driver for the relatively higher heavy metal concentrations seen for urban vs. rural farms in this study, supported by past UK research (Alloway, 2004; Giusti, 2011; Ander *et al.*, 2013; Crispo *et al.*, 2021). Additionally, the locations of the urbanised farms are in regions and cities with a history of prior mining and industrial activity, particularly in the southwest and north of England (Norton *et al.*, 2015). Lead mining in particular was common in the area near U-O-2 (Giusti, 2011), which likely explains its high Pb levels (Appendix H, Table 83).

One rural farm located in southwest England (R-C-4) emerged as a major outlier in this dataset, having the highest soil concentrations of As, Pb, Cu, Fe, and Zn compared to all other farms (Appendix H, Table 83). For As in particular, concentrations are one to two orders of magnitude higher than seen on other farms (80-150 mg kg^{-1} vs. <20 mg kg^{-1}), but similar to some of the outlier values found the topsoils of Bristol, one of the major post-industrial cities in southwest England, where this farm is located (Giusti, 2011). The fact that heavy metal concentrations are relatively high for both cultivated *and* undisturbed soils on this farm (albeit higher for cultivated) indicates the influence of other drivers besides agricultural activity. One of the main soil parent materials in this area (sandstone) is known to be high in As, thus contributing to higher ‘baseline’ soil As concentrations in this region (Giusti, 2011). Mining activity can also contribute to the mineralisation of parent materials

(Ander *et al.*, 2013), with further contributions from mining industrial emissions and waste. Thus, it is likely that the main contributor to the relatively high heavy metal concentrations seen on R-C-4 is the extensive mining activities that took place in southwest England from approximately 1860-1900 (Norton *et al.*, 2015). During this time, this region was a major world producer of arsenic and was also known for mining minerals containing Cu, Pb, and Fe (Farago *et al.*, 1997). Soils in this area have been cited as being contaminated with As and Cu as a result of this mining and smelting activity, with much of this contaminated land being agricultural (Thornton, 1994; Farago *et al.*, 1997). Other studies investigating soils in prior mining areas have also highlighted As and Pb as major soil contaminants, with high levels of Fe also found (Gonzalez-Fernandez *et al.*, 2011). Thus, the heavy metal concentrations on this farm can be related mostly to this prior industrial activity, as with the urban farms.

Even though industrial activities are seen to be more prolific drivers of heavy metal emissions in many cases, agricultural activities can and do contribute to heavy metal accumulation in soils through the continued application of inputs that contain heavy metals (Nicholson *et al.*, 2003; Smith, 2009). This is evidenced in this study as most heavy metal concentrations are higher in cultivated soils versus undisturbed soils on the same farm (Appendix H, Table 83). Sources of heavy metal inputs on farms include livestock manures, inorganic fertilisers (mainly phosphate fertilisers), lime, pesticides and other agrochemicals, irrigation water, composts, sewage sludge, and other industrial wastes that may be applied to agricultural soils (Nicholson *et al.*, 2003; Otero *et al.*, 2005).

The heavy metal concentrations in these inputs vary depending on the type and source. Phosphate fertilisers and livestock manures are important sources Cd and Zn, with the prior also contributing to Cu (Nicholson *et al.*, 2003). The application of sewage sludge as a fertiliser (restricted in organic agriculture) has also been seen to increase soil Zn, Cu, Cd, and Pb levels (Berrow and Webber, 1972; Chander and Brookes, 1991; Smith, 2009). This practice may explain the relatively high levels of Pb seen on R-C-3, as the spreading of industrial wastes and sludge from nearby cities was historically common for agricultural land in this area; otherwise, the largest Pb input in UK agricultural soils usually comes from atmospheric deposition (78%) (Nicholson *et al.*, 2003). Thus, for most other farms, Pb levels are likely more related to other industrial processes in the area (and resulting atmospheric deposition from emissions) rather than agricultural activity (Nicholson *et al.*, 2003). However, compost can also be a source of Pb, as well as other metals such as such as Zn and Cu (Pinamonti *et al.*, 1997; Smith, 2009).

Accumulated heavy (or otherwise potentially toxic) metals in soils do not always pose a risk. However, if levels become too high, this can lead to phytotoxicity, disruption of soil microbial processes, and also potential health implications for animals and people if exposed from the soil or through contaminated crops (Chaney, Sterret and Mielke, 1984; Nicholson *et al.*, 2003; Antisari *et al.*, 2015). In terms of human health risks, exposure to toxic levels of As has been associated with vascular disorders and various cancers (Mohammed Abdul *et al.*, 2015); Cd with kidney and liver disease (Nogawa *et al.*, 2004; Hyder *et al.*, 2013); Pb with neurological damage (Sharma, Chambial and Shukla, 2015); and Zn with potential reproductive effects (Nasiadek *et al.*, 2020). Compared to these metals, Fe is generally considered to be less toxic and thus less of a health risk (Gonzalez-Fernandez *et al.*, 2011). However, excess levels of Fe intake over the long-term may present a risk, particularly for individuals with disrupted iron metabolism, such as those with hemosiderosis (Gonzalez-

Fernandez *et al.*, 2011). High levels of soil Fe can also be concerning for crop production if this results in the development of an iron pan, which prevents root penetration (Cunningham, Collins and Cummins, 2001).

To indicate whether the metal concentrations found in the soils in this study are a potential cause for concern, values are compared to 'normal background concentrations' (NBCs) for English soils. NBCs are meant to typify normal levels of heavy metals that do not pose an unacceptable risk to human health or the environment, defined in relation to potential site contamination rather than specifically for agricultural activity (Defra, 2012; Ander *et al.*, 2013). The NBCs for English soils, as defined by Ander et al. (2013), are categorised for different areas in England that might be influenced by different sources (different 'domains'), such as areas with certain types of soil parent material or urban areas. The NBCs represent the upper 95% confidence limit of the 95th percentile for the soil results associated with a particular domain, such that they represent the highest concentration of a contaminant that is likely to come from normal background reasons for that particular domain (e.g., from urban contributions for the urban domain).

Out of the heavy metals discussed within the results of this chapter, only Cd, Cu, and Pb have defined NBCs for the urban domain (respectively 2.1, 190, and 820 mg kg^{-1}), thus highlighting that urban activities are a major driver of differences in heavy metal concentrations between different English soils (Ander *et al.*, 2013). Comparatively, the mean and median concentrations for these heavy metals in the urban soils are all much lower than the urban NBCs at both depths. Concentrations as assessed on individual urbanised farms are also all lower, with the exception of a slightly higher Cd concentration (2.95 mg kg^{-1}) seen in the undisturbed soil samples from PU-O-2 (Appendix H, Table 83). Finally, since no value for the urban domain was defined for As, values have been compared to that for the principal domain (32 mg kg^{-1}), which includes all other data points not defined in other domains of interest (Ander *et al.*, 2013). Again, values from all urbanised farms were lower than this NBC.

The metal concentrations from urban farm soils are also compared to values previously reported for UK urban horticulture sites to provide additional context (Crispo *et al.*, 2021). In this case, values for As and Cd are similar, but median values for Cu, Pb, and Zn are approximately half of those previously reported, and even mean values are lower. This may be explained in part by poor recovery of certain heavy metals (Table 60), particularly As and Cu, with recovery rates indicating that the concentrations of these heavy metals may potentially be up to 2x higher. Another explanation could be the different landscapes sampled between the studies (allotment soils vs. various urban farm landscapes). Crispo *et al*. (2021) noted that pesticides, paint particles, bonfires, rubber tyre abrasion, and runoff from metal surfaces (e.g., gutters and rooves) can all be contributors to As, Cd, Pb and Zn concentrations; however, most of these are more relevant for allotments in or near residential areas than the urbanised farms in this study, which are not very close to residential buildings. Median metal concentrations in urbanised soils are also similar to values reported for arable and horticultural soils nationally, except for Pb, where urbanised soils have nearly double the concentration (82 vs. 42 mg kg^{-1}, respectively).

Concentrations in rural soils are compared to NBCs for the principal domain. Both median and mean metal concentrations from rural soils are lower than the principal domain NBCs, as defined for As (32 mg kg^{-1}), Cd (1 mg kg^{-1}), Cu (62 mg kg^{-1}), and Pb (180 mg kg^{-1}). However, when considered individually, As concentrations for both cultivated and undisturbed soils on R-C-4 exceed the NBC, with Cu concentrations in cultivated soils also exceeding the NBC. However, a separate domain for As and Cu has been defined for soils affected by non-ferrous mineralisation (and associated mining activities), which include soils in the southwest of England, where this farm is located. This further supports the hypothesis that the higher metal concentrations on this farm are a result of mineralisation of bedrock naturally high in these metals and also mining activities. Thus, when comparing to mineralisation domain NBCs as defined for As (290 mg kg^{-1}) and Cu (340 mg kg^{-1}), concentrations in R-C-4's soils are lower. Median heavy metal concentrations from rural soils are also mostly similar to those found for arable and horticultural soils in the Countryside Survey, although concentrations of Cd and Zn are slightly lower than national values by 35 and 45%, respectively (Emmett *et al.*, 2010).

For the most part, metal concentrations in the soils investigated in this study are not higher than what would be expected from similar English soils, and thus do not likely pose a large concern for contamination. However, this must also be considered with respect to the relatively low recovery for some metals (As and Cd); if these values are actually higher, based on the recovery rates, then the levels on some farms, particularly R-C-4, may be a cause for concern.

Additionally, although the NBCs provide a suitable reference for any extreme heavy metal concentrations within a particular domain (Ander *et al.*, 2013), whether or not elevated levels seen in certain domains (e.g., urban) will result in environmental or human harm is less clear. Of particular interest is the bioavailable and mobile fractions of total heavy metal concentrations, as they are more related to the propensity for the metal to be taken up by the plant or to be leached, respectively (Violante *et al.*, 2010). The bioavailability of heavy metals is influenced by soil pH and also soil carbon content, due to potential immobilisation via OM (Thums, Farago and Thornton, 2008). Indeed, although phosphate fertilisers, limes, and composts contribute heavy metals inputs to the soil, they are also known to immobilise heavy metals and thus reduce bioavailability and mobility (Bolan and Duraisamy, 2003). Therefore, total heavy metal concentration may not adequately represent the fraction that is a current health or environmental risk (Iwegbue *et al.*, 2007).

Previous research investigating bioavailable heavy metal fractions in allotment sites across 10 UK cities found that this fraction represents only a small part of total heavy metal concentration (0.1 %–1.8 %), indicating a low risk of crop uptake (Crispo *et al.*, 2021). On the other hand, Entwistle et al. (2019) found high Pb bioavailability (59%) for urban gardens in Newcastle, which had total Pb levels exceeding UK Governmental screening levels for allotment sites; however, despite these levels, they found that Pb concentrations in the blood of the gardeners and their non-gardening neighbours were not statistically different. Thus, although heavy metal bioavailability is generally regarded as a better indicator for the potential crop contamination (Ge et al., 2000), even this may not be directly relatable to health impacts, which are also highly context-dependent. Additionally, the bioavailability tested at one point in time does not necessarily mean that higher fractions will not become available in the future (Iwegbue *et al.*, 2007). Overall, this points to the importance of testing any soils used for crop production for heavy metal concentrations and bioavailability, particularly in urban areas, to identify and reduce any environmental and health risks.

4.2.3 Limitations and future research

Although this research provides crucial insights to the soil health status of different local farming systems across England, there are limitations to this study. For one, this study included a limited number of sampling locations, evaluating just ten farms overall based on those also studied in the lifecycle assessments of the previous chapter. The limited number of farms and the fact that these farms existed across different regions of England, with different climatic conditions, soil types, and historic land management, means that this data should not be used to draw wide-ranging conclusions about the soil health of different farm management types or different levels of urbanisation. However, in line with the aim of this study, this data *can* be used to provide context to the types of ecosystem services that may be provided by different types of local farming models, in particular highlighting the many services (or disservices) that contribute to farm sustainability, but that are often not considered when using only carbon accounting or lifecycle assessment methods.

An additional limitation of this research is that only a subset of soil health metrics is evaluated. As one of the main listed advantages of organic farming is increased biodiversity (both in and above the soil) (Maeder *et al.*, 2002; Lotter, 2003; Tully and McAskill, 2020), future research would be improved by including biological soil health indicators and biodiversity measures. Biological soil indicators like microbial biomass are especially important in the context of soil resiliency (Lehman *et al.*, 2015); thus, farming systems that support microbial growth and diversity also support more resilient local food systems. This research also only examined total soil nutrient and heavy metal concentrations, which provide an overview of soil nutrient stocks and potential contamination risks, but do not provide specific information about availability of nutrients to plants (important to identify potential deficiencies) or the solubility and mobility of nutrients and metals (important to identify contamination risks from leaching and runoff). Thus, future research would also be improved by examining fractionation of these elements (Emmett *et al.*, 2010; Violante *et al.*, 2010; Darch *et al.*, 2014). Finally, in order to better understand the potential for different farming systems to contribute to carbon sequestration and thus climate change mitigation, SOC could be measured to lower soil depths, such as 100 cm (Gregory *et al.*, 2011; Edmondson *et al.*, 2012), to provide a better indication of carbon storage in slower soil pools and how this may be affected by different agricultural management types (Stockmann *et al.*, 2013). Assessing this in farms over time would also help determine if and when changes in management practices are able to change long-term SOC storage at lower soil depths (Johnston, Poulton and Coleman, 2009).

4.3 Chapter conclusion

This study aimed to evaluate soil health metrics across different types of local horticulture farms in England to provide further context to the types of soil-based ecosystem services that can be found on farms, which are not accounted for within the lifecycle assessments of Chapter 3. The results show that uncultivated and organically-managed soils across the study farms had lower bulk densities, higher water-holding capacities, and higher SOC concentrations than conventionally-managed soils, thus contributing more positively to the ecosystem services of water retention and flood mitigation, erosion control, improved soil structure, and carbon storage. Considering that the lifecycle assessments of Chapter 3 found relatively high environmental impacts on many urban and peri-urban organic farms, mainly associated with large additions of compost, this study provides additional context by highlighting that there are also many ecosystem benefits that come from these farms, which are actually supported by higher organic matter additions and thus higher SOC. On the other hand, potential soil degradation is observed on some of the conventional farms, particularly related to soil compaction and SOC depletion, which may threaten the sustainability and resiliency of these farms in the long-term. Still, other conventional farms were able to maintain better soil health conditions, likely due to certain management practices (e.g., minimum tillage and cover cropping), showing that the management decisions made by organic and conventional farmers alike can provide key ecosystem benefits. The protection of uncultivated spaces on farms (e.g., hedgerows, woodlands, field margins) is also key to ecosystem service provisioning in agricultural areas. Finally, the evaluation of potentially toxic metal concentrations in this study showed relatively high concentrations for urbanised farms, as well as for certain farms in areas with historic mining activities, pointing to potential risks for plant, animal, and human health; however, all levels were below the normal background concentrations for their respective sources in England, indicating that this risk is likely low.

Overall, this study shows that farm sustainability cannot be simply evaluated through environmental impact assessments (such as LCA) on crop production areas; other unproductive spaces on farms, as well as other ecosystem services, must also be considered for their role in farm sustainability. However, environmental ecosystem services like those evaluated in this study also do not provide the full picture. There are also many social and cultural services that can be provided by farms. Although these services were not measured within this thesis, many farmers provided examples colloquially, such as donating to local community groups. As the main purpose of farming is generally seen as producing food to feed the human population, the contribution that different local farming models are able to make to food security is also an important indication of the role that these models play in a sustainable and resilient local food system. Food security is related not just to farming systems, but also to the food supply chains in which they sit. Although much of food and agriculture research today focuses on how to make farms collectively more sustainable and productive, an equally (if not moreso) crucial agenda is to find ways to reduce food waste and improve food access so that the food already being produced is fully utilised and reaches all who need it. Thus, the final study in this thesis explores issues of food insecurity as explained by those experiencing them, examined during a time of crisis that threatened food systems across the globe.

Chapter 5: Experiences of food insecurity and community responses during COVID-19: A case study in the UK

This results & discussion chapter is structured as follows. First, a short description of Foodhall's emergency food response during the first COVID-19 lockdown is provided, along with quantitative data about the scope of this response. The results then move to a more detailed portrayal of the lived experience of food insecurity during COVID-19, gathered through interviews with fourteen individuals receiving emergency food support from Foodhall. In the discussion section, the major concepts that inductively arose through the stories of participants are linked with theories and topics that have been presented within this the introduction for this chapter (see: Section 1.1). The chapter concludes by summarising the research and recommending strategies and policies toward supporting community resilience.

5.1 Results: Foodhall's COVID-19 response

This research focuses on the activity of one community food project in Sheffield, England – the Foodhall project – as a key case study of community responses during the COVID-19 pandemic. More detail about the aims and activities of this project before the pandemic have been provided in Section 2.3.1.

During the very first week of the UK COVID-19 lockdown (23 March 2020), the Foodhall project (hereafter referred to as 'Foodhall') shifted from operating as a community café to providing a city-wide food delivery service - something the project had never done before. The project began delivering free meals, food items, and other essentials directly to people's homes. A warehouse was built to store and pack food, and a helpline service was created for people to call to request a food parcel. Volunteers were recruited to create delivery maps and organize routes with different volunteer drivers and cyclists. Food donations were sourced widely from supermarkets, restaurants, local farms, and the city council. Because of the non-hierarchical and community-based nature of the project, Foodhall was able to act quickly to respond to an emergency that saw foodbanks and other emergency sources of food shut down, at a time when more people than ever lacked access to food due to reduced income and mobility.

Foodhall's COVID food parcel service included both free deliveries directly to people's homes (available to all postcodes in Sheffield) as well as a take-away, front-of-house service, to support those who did not have an address to receive deliveries. The service was open to anyone in Sheffield and required no means testing or referrals – meaning that anyone could receive a free food parcel with no questions asked. In this way, Foodhall aimed to support those who did not qualify for, or could not access, other food banks or government and council programs. Anyone requesting a food parcel would receive a delivery the same or next day, with deliveries happening 5-6 days a week. Each person requesting a food parcel would have a personal call with a Foodhall volunteer, to provide details on the number of people in the household requiring food and other essentials (e.g., toilet roll, diapers, shampoo, etc.); any dietary requirements; and any specific foods requested due to available cooking facilities (e.g., frozen meals, microwavable meals, canned goods, etc.). As this was intended as a short-term service, there were no options for receiving repeat parcels on a certain day; however, recipients were encouraged to call or visit again anytime they needed another food parcel, and indeed many visited the physical location several times a week or called the helpline once per week or fortnight. Finally, trained counsellors employed by Foodhall also provided wellbeing check-in calls to delivery recipients who expressed interest.

Foodhall's service began the very first week of UK lockdown (23 March 2020) and continued at maximum capacity until the end of July, when deliveries temporarily paused as the project moved buildings. During the first week of lockdown, the project fed 70 people in total, through the deliveries and front-of-house service. After one month, the delivery service reached full capacity, based on the number of volunteers able to be in the space (following lockdown guidelines) and the amount of food available. From the 20th of April to the end of July 2020, the project was delivering, on average, 168 parcels per week and feeding a total of 865 people per week, with approximately 55% of these people receiving deliveries and 45% receiving parcels through the front-of-house. Children comprised 35% of the people being fed through deliveries. This service relied on a total of 473 volunteer hours per week.

Through mapping efforts undertaken by a Foodhall volunteer, it was found that during the first 10 weeks of food parcel delivery (23 March - 1 June), nearly 75% of food parcels were delivered to households which were located within the 20% most deprived areas in the country. From June to July 2020, a local non-profit organisation called Voluntary Action Sheffield began collecting data from 21 organisations across 28 different sites providing food support across Sheffield (not including government parcels and several smaller organisations). Based on this data, Foodhall was identified as one of the largest emergency food responses in Sheffield, feeding on average 20% of the total 4,770 people fed per week by all 21 organisations providing data. Indeed, after recognising the role the project was playing in the city's emergency food response, the city council began signposting people to Foodhall as a first option for emergency food relief, and also began providing Foodhall with food donations (for a limited time) to keep the project operational.

The fact that these community organisations were feeding such a high proportion of people showed the immense need for community food infrastructure during the pandemic. Since these statistics did not include people receiving government food parcels, this shows how many people were in need but not supported by the nation-wide food support strategy. Indeed, the fact that Foodhall was mainly delivering food parcels to people living in the most deprived areas of the country, as well as providing food to many people without housing, showed that those living on the edge were not getting the support they needed from national measures. As people 'fell through the cracks' of state support, community efforts rose to respond to local challenges and needs. This included not only formal community organisations and charities, but also informal, neighbourhood-based groups (called mutual aid groups) that sprouted up across cities and villages in the UK to provide help and support to those in need, especially for those self-isolating and unable to physically obtain food, medical prescriptions, and other essential items (Fernandes-Jesus *et al.*, 2021; Ntontis *et al.*, 2022). This showed how social support structures, and therefore a sense of community to initiate these structures, were necessary to keep people alive, fed, and cared for.

By August 2020, lockdown restrictions began to ease, as restaurants, pubs, and other community centres were allowed to re-open. Foodhall paused their delivery service at the start of August, as the project was moving buildings. Foodhall's service began again mid-

August, although at a much-reduced capacity. The project aimed to re-focus on opening their café and social eating space, and thus began calling delivery recipients individually to signpost them to other (now open) community projects. Throughout 2022, the project still continued to provide food deliveries, although only through referrals by other community organisations in Sheffield. However, at the end of 2022, this project unfortunately closed after seven years in operation, due to the financial toll the project endured during COVID.

The author of this paper, as a former Foodhall volunteer and researcher at the University of Sheffield, aimed to capture the stories of those relying on emergency food support during the pandemic. The narratives of the fourteen individuals that were interviewed reveal the multi-dimensional barriers to accessing food during COVID-19 and show the integral role that community food infrastructure can play in crisis response.

5.2 Results: The lived experience of food insecurity during the COVID-19 pandemic

During the first months of the COVID-19 pandemic, study participants faced barriers to achieving all four dimensions of food security, including: financial and physical access to food; local availability of sufficiently nutritious food; utilisation or capacity issues; and consistency of the food supply. This section details how these challenges emerged and how they were interconnected, and often multiple, for participants. In particular, I highlight the individual, social, and place-based resources necessary to achieve food security, how these influence each other, and how these were constrained during COVID. Drawing on the words used by participants, three overarching and interrelated themes emerged from the analysis: the stretching of resources, the struggle to cope, and exacerbated emotional and physical stress.

These three themes are explored throughout the first three qualitative result sections. As household resources and capabilities were stretched, this resulted in new direct barriers to achieving food security, related to financial struggle, self-isolation requirements, and change of daily life during COVID. Participants thus scrambled to find ways to cope with newfound struggle, but at the same time the options available were diminished as many traditional support options became inaccessible and participants' personal networks were also stretched. The resulting stress, feelings of lost independence, self-stigmatisation, social isolation, and mental health challenges became a further barrier to achieving food security as this often prevented people from asking for and accepting support. Stigma and isolation were embedded within the ways that participants internalised their burdens, viewing their struggle as a personal failure, characteristic of the individualistic neoliberal-ableist ideal, despite the fact that so many were also struggling at the time.

Finally, the last two results sections bring to light what participants expressed helped them most during this time – people. Participants highlighted how being able to share about their struggle with others and, in doing so, recognising that they were not the only ones struggling, eased the mental burden and made them feel more comfortable accepting help from others. This was mediated by the ability to engage in reciprocal relationships, thus providing an option for exchange that circumvented the stigma of accepting a "handout." With reciprocity at its foundation, the concept of mutual aid emerged, and we see how participants cared for others and were cared for by their personal networks and communities. In the final section, I explore how community spaces like Foodhall play a vital role as a connector – bringing people together when times are hard, and building connections between local groups, businesses, and public bodies to facilitate a rapid crisis response that caters to local and place-based needs. Incorporating community food organisations like Foodhall as a critical part of a city’s food infrastructure – and thus creating a community food infrastructure – builds resiliency as these spaces can quickly adapt to provide logistical support to people during crisis, thus easing the burden so people can address other life challenges. However, the flexible and grassroots nature of these organisations is set against issues in maintaining the consistency of these spaces in the long-term, which would require further state support.

5.2.1 Stretched resources: Barriers to accessing food

The COVID-19 pandemic, with the resulting lockdown restrictions and social distancing guidelines, significantly changed everyday life for people in the UK. People experienced increased barriers to access basic goods and services to meet their needs. In regards to food, the pandemic created a group of newly food insecure people and also deepened the hardship of accessing food for those who were already food insecure before the pandemic. In this study, eight participants had not been using any form of food support before the pandemic, and out of these, six had never used food support ever before in their life. Participants reported that the need to reach out for food support was spurred by two direct barriers to accessing food during the pandemic: financial and physical access.

In this study, increased financial strain was the main barrier identified by participants, as they struggled to afford food. Eleven out of fourteen participants reported financial strain as having impacted their ability to afford food for themselves and their families during the pandemic. Out of these eleven, five participants expressed that they had not been struggling financially before the pandemic, but that now their financial situation had been negatively impacted due to COVID-19. The other six confirmed that they had been struggling financially before COVID, but that the pandemic had increased their household expenses. Participants indicated that the financial strains that came with lockdown were related to: changes in personal or household income, related to job loss, self-employed income loss, or benefit loss; changes in household composition, which also affected household income; a rise in household utility costs; and a need to spend more money on food. In addition to financial barriers, half of the participants were classified as clinically vulnerable to the virus and were therefore shielding (not leaving the house). This created a direct physical barrier to accessing food, which compounded financial struggles.

Despite the fact that those classified as clinically vulnerable were eligible for priority delivery slots from supermarkets, many participants in this situation reported that they were unable to access these and thus were struggling to physically obtain food without leaving the house. These participants had to rely on others to shop for them, including neighbours, other family members, and local support groups, such as National Health Service (NHS) volunteers. In some cases, participants had to resort to paying someone to shop for them. In addition, participants were often only able to go to or send people to their local shops, thus unable to access the larger, lower-cost supermarkets located farther away due to the combined issues of self-isolating and the inability to afford or use public transport. Even in local shops, the lack of available food often meant budget-brand items disappeared first, limiting the local availability of affordable food choices. These factors further affected participants' already strained financial situations by increasing the cost of food.

Diane shows how changes in her living situation and the inability to physically go out and get food impacted her financial situation. As a recently displaced domestic abuse survivor, Diane was unable to leave the women's hostel where she was living because of the need to shield before and after a surgical operation. As she was now living in a new part of town without a support network nearby, Diane explained that her only option was to send National Health Service (NHS) volunteers to local shops, which were more expensive than the larger supermarkets where she would normally shop. The need to rely on volunteers thus reduced her choices and increased her food costs.

> I'm struggling to get food. I've had to spend quite a bit at my local shop. I've often sent people [NHS volunteers] to the local shop, which costs three times more than it does in the supermarket. (Diane, age 40-50)

Diane's barrier to physically accessing food exerted further strain on her already limited and inadequate Universal Credit income. The lack of affordable food in her area further affected her financial resources. However, the presence of another local resource – NHS volunteers – meant that Diane did have someone she could ask to shop for her.

As with Diane, many found themselves in a new or altered living situation during lockdown. With the announcement of lockdown, some individuals moved in with family or were instead separated from their families, not knowing when they would be able to move again. This complicated household dynamics, income, and costs. It also meant a change in household capabilities, including the skills and services people in the household were able to provide for one another. Thus, the ways that households were reorganised just prior or during the pandemic created new financial and physical barriers to accessing food, as well as changing how people were able to address these barriers.

Caroline's story provides an example of how changes in the household affected her ability to afford food. As an older woman surviving off her pension, Caroline began supporting her two adult sons during lockdown. Her sons, who were previously self-employed, had now found themselves in the gap between loss of income and receipt of benefits. She describes in detail how they had to stretch resources between the family, with her sons even sharing fuel by "siphoning diesel out of each other's vehicle depending on who's going out." Caroline's pension became the primary source of income to support three adults, whilst her utility costs increased by four times compared to what she usually paid at that time of the year.

> I'm the only one that's really got a steady income, if you want to put it that way, coming in. My pension every week. To keep us afloat. Well, to keep the house afloat and the food coming in if you know what I mean. In a way, I'm supporting my sons, and it's obviously the gas and electric. We've used more of that while they've been here because it's been 24/7 use of electric. My electric bill, my gas bill has gone up. (Caroline, age 70-80)

Caroline's economic strains came about from having to support an increased number of people in her home with what was previously an adequate income. The political-economic context that created an absence of other (or enough) support for her adult sons placed the burden of responsibility onto her.

Others found themselves living on their own during lockdown, which impacted the financial, geographical, and knowledge-based resources that had previously been available to them from others living in the household. The physical ability to shop and cook for oneself became a significant issue during the pandemic, especially for those who were unable to leave their homes and did not have others to help them.

For Kathy, changes in her household composition meant a loss of vital household capabilities. Although she did not consider herself food insecure before the pandemic, the onset of lockdown pushed Kathy into a severe state of food insecurity as she found herself living alone. Kathy had previously suffered from a brain haemorrhage, and for some time, her adult son was her carer. He received a carer's allowance that, along with Kathy's Employment and Support Allowance, helped support them both financially. Just before lockdown, Kathy's son moved out, resulting in a loss of the carer's allowance. This loss left Kathy struggling to stretch her income to cover the same household expenses. In addition to losing this income, she had to pay someone to go shopping for her, as she could not manage this with her disability. What had previously been a household capability now became an additional cost upon an already lowered household income. Atop of this, Kathy's problems were also compounded by other family members who were not living with her. As many were struggling at the time, this led to tense and dire situations.

> My other son, he got arrested. He took my bank card with him and emptied all my bank account. I had no money. Practically my word against his. He's in prison now. I don't know what's going on… I had a brain haemorrhage. I keep my bleeding PIN number with my card, which is stupid, isn't it? (Kathy, age 50-60)

Kathy's severe financial struggle meant that she could not afford internet and phone service most of the time and was often without food; in some cases, she had to resort to eating out of bins when she could not call for an emergency food parcel or pay someone to get her shopping. It becomes clear that her direct barriers to accessing food – financial and physical access – were also linked to physical and mental capabilities as well as family and place-based resources (e.g., availability of affordable food in local shops and the presence of local volunteers to do the shopping). Although Kathy tried to find help to sort out the loss of the carer's allowance in the household, this was constrained by the fact that she often did not have phone credit and struggled with her memory. Her son moving out meant both a loss of finances but also capabilities. Thus, Kathy's story shows how food insecurity is not an isolated situation but is linked to the circumstances of her household and other family members.

In addition to struggles accessing food, changes in household configurations also impacted people's ability to utilise food. An example is Liam, who began living on his own during the pandemic for the first time in over twenty-five years. Liam struggled with an alcohol addiction which got worse when lockdown began, eventually resulting in his partner telling him to leave the family home. This diminished his access to food knowledge and budgeting techniques held by his partner (e.g., knowing how to cook with available items). He explains:

> [Before lockdown] I was living with my partner. We were together six years and she sorted everything out, money-wise. Then before that I was with my ex-ex-partner for 17 years. Again, I just let things get done. Now, I've been chucked here, not having a

> clue about bills. About anything, what I need to pay. What's priority. Just head up my butt sort of thing. (Liam, age 40-50)

In addition to the difficulties in managing his own finances, the context of living in emergency accommodation, relying on food support, and needing to make meals for himself stretched his food skills. Having to rely on others for food (e.g., Foodhall’s food parcels) meant that he wasn’t able to plan meals ahead. Further, his emergency accommodation did not have an adequately-sized freezer, meaning he was unable to store food items for the long-term, further reducing his ability to budget and plan ahead (e.g., by buying and storing sale items). He explained that at one time, he ended up with “nine tins of tomatoes” from the food parcels, because he felt like he couldn’t use them in any dishes without having any minced meat to make a spaghetti bolognese. Sometimes, situations like these even caused him to skip meals:

> There are some days that I’ve just had breakfast and nothing else. It’s because there’s things in, but it’s not as though you can do much with them together. One meal I had, I had a tin of tomatoes, mushy peas, garden peas, and you’re just thinking, well they don’t go together, do they? What are you going to make with that? You’re still having to go out and try and buy little bits of stuff that you can actually just make a meal with. (Liam, age 40-50)

Liam does not have a complete lack of food knowledge – indeed, he understands what ‘goes together.’ However, he often did not know how to cook meals with the combinations of food items he received. There were few opportunities for developing budgeting, cooking, and meal planning skills, as many community centres had stopped classes or closed during the lockdown. The loss of his partner's food skills further diminished his options. Combined with a lack of income and support, this undermined his overall ability to achieve food security.

These stories illustrate that food security is linked not only to personal capabilities, but also to the capabilities and resources available within one’s family or household. Family resources can expand the ability to achieve food security when capabilities and resources are shared, but also limit them when those people are lost or are also struggling.

It is clear that some participants were already struggling to access food before COVID. However, circumstances of the pandemic further shrank the resources people had available to them to obtain, afford, and utilise food. This was seen in the ways that COVID altered household circumstances, which stretched individual resources and made social networks unavailable or problematic for support. The direct financial and physical barriers to accessing food were interwoven with other struggles that compounded people’s ability to cope with food insecurity. This is why simple cash payments or food deliveries did not completely erase people’s struggles. The resources needed for individuals to have the capability to achieve food security was linked to geographical context, health, family life, and available social support networks; changes in these resources due to COVID contributed to new barriers in accessing food and their ability to cope with hardship.

5.2.2 Struggling: Coping with food insecurity

With their resources stretched, trying to overcome the mounting financial and physical barriers to accessing food, participants had to find new strategies to support themselves. For those who had been food insecure before the pandemic, new difficulties arose as many coping strategies used previously were no longer viable. The newly food insecure had to scramble to navigate a situation they had never before experienced. Being able to actually find the help and support that people were eligible for was a challenge faced by many, with local services closed, government services stretched, and limited phone and internet access.

This section discusses the strategies used by participants to cope with food insecurity and the additional barriers they faced in finding help and support, largely due to COVID-19. Participants first adopted strategies within their own resource capacity, such as 'trading down' and 'going without.' Meticulous budgeting and meal planning, often not employed before the pandemic, became essential to ensure that participants would be able to feed themselves and their families. When these strategies were not enough, participants turned to informal social networks if they existed and then to external support, such as government programmes and local services. Participants found many government programmes difficult to access even if they were eligible for them, which left many struggling to find new local support services. We see that many of the coping strategies available to our participants were only temporary solutions, which could not provide a means to ensure consistent and stable food security in the long-term. In fact, some of the strategies adopted by participants could be particularly damaging to the long-term capability to achieve food security by impacting their physical and mental health.

5.2.2.1 Individual coping strategies

As resources became stretched and poverty levels increased during COVID-19, people first sought ways to manage this themselves. Despite the rhetoric at the time of everyone 'being in it together' and needing to work together to 'save lives' and 'protect the National Health Service' (Department of Health and Social Care, 2021), we see that our participants framed their struggles as personal, individual experiences, characteristic of neoliberalism. Participants were organised, creating meticulous budgets and explaining their calculations for food, rent, utilities, and transport in detail. People cancelled internet, TV, and phone services and also cut back on food or resorted to buying cheaper foods to reduce costs. These budgeting decisions revealed what people valued and how diminished their choice sets for achieving food security had become.

Often, the difficult choices people had to make between their various needs were calculated and deliberate. Linda, an American citizen living in a village on the outskirts of Sheffield, described the difficult situation she was placed in during the first UK lockdown. After leaving an abusive relationship in the U.S. the previous year, she had come back to the UK, where she had lived previously, for a fresh start. She found herself unable to work during COVID as her previous place of employment closed with lockdown, and at the same time, her visa expired. As she had been working as a self-employed contract worker, she was also ineligible to receive furlough payments. She relied solely on U.S. Social Security payments for income, explaining the trade-offs she had to make between her needs and her after-rent monthly income of £147 per month, which often left her with very little money for food.

> With my [U.S.][1] social security, what I get is 747 dollars a month, that's about 622 pounds. So, the 475 for rent, and I strictly budgeted top-up cards for my gas, my heat,

[1] Throughout the quotes in this chapter, square brackets enclose words which have been added to clarify meaning and provide a brief explanation.

> my phone, very, very carefully. There just wasn't anything for food and then there wasn't - you know, I could budget taking the bus once in a while, but I couldn't budget taking a bus to go get food. (Linda, age 40-50)

Crisis disrupted Linda's capability to be food secure, and the lack of available and affordable food in her area meant that she had to rely on other capabilities to stretch her finances and maximise food use. Linda was extremely organised, keeping detailed budgets and planning meals well in advance. She described how she stretched out one tin of ham or one curry she received from Foodhall into multiple meals. Often going without meat, which she missed, Linda adjusted to cooking mainly vegetarian, low-fat meals that "go very, very far." Linda even used this opportunity to focus on improving her physical health, as she was overweight, losing over 20 kilograms (44 lbs) "by design."

> You can't just buy a frozen thing that doesn't go that far. My partner's always shocked, he's like you're so organised. Well, you know, if you're super organised and you do everything from scratch, and then you have it pre-prepared, I whip together a meal really quickly. But it's because I've done all the prep work. It's not because I've bought it ready-made. (Linda, age 40-50)

Careful meal planning, aided by prior food and cooking knowledge, was also an important coping strategy used by participants, which required both skills and time. Often, this was not part of their food routine before the pandemic. Linda described how before the pandemic, she didn't think too much about planning meals and was just able to get whatever she wanted from the shop. Similarly, Caroline described that what she and her sons ate before the pandemic was "spontaneous" but that she now carefully budgets to plan one "proper family meal" every day for the three of them.

> We have been able to manage from one weekend to the next. Just careful budgeting, what I buy, what you [Foodhall] provide. It's sort of, 'That'll go with that, and we've got that in the cupboard. I can go and buy such and such a thing', and then it's planning a week's meal. One meal a day and then the rest of the time, it's 'you'll have to go and help yourself to toast or have some beans on toast or have a tin of soup.'" (Caroline, age 70-80)

Linda and Caroline's careful organisation and budgeting strategies demonstrate a clear ability to utilise food and plan ahead, but these strategies were still not enough to stretch their income and overcome the lack of affordable food access and availability in their local areas.

As participants made choices and trade-offs between their various needs, we see that food strategies were embedded within a wider context of what could and could not be sacrificed. Having food often meant going without something else. Liam described these painful calculations, where having the food he wanted was set against being able to see his children.

> Meat's not cheap, is it, either? When I'm trying to balance my money, I'll go in and I'll see something, and I'll think - it's only £5. But then I'm thinking, bloody hell, that's £5 that could go into petrol and get me up and see my kids for probably two days. I'm always trying to think, is it worth it just to have a nice meal or do I use it and save it so I can see my kids? They win every time. (Liam, age 40-50)

Randy (age 50-60) similarly skipped meals so that he could afford rent and feed his daughter and wife who has a disability, because he felt their needs were more significant than his. He said, "I had to pay my rent before getting food. I thought the rent were more important than getting food."

These stories demonstrate that the calculations people made were often not choices at all. For some, like Linda, keeping a phone meant accessing government services for help with her visa and connecting to family and friends, but the cost was being unable to travel to a large supermarket to purchase cheaper food. For others, like Liam, it was a choice between maintaining family relationships and eating. For Caroline and Randy, going without was set against the ability of family members to manage and have a place to live.

It becomes clear that as participants had to prioritise their various needs and the needs of their families, food was one area where significant trade-offs were made. This was reflected in how diets changed considerably for participants during the COVID-19 lockdown. Linda, Caroline, and Liam specifically mentioned foods they used to enjoy but now could no longer afford, such as meat or take-away meals. Although Linda was able to use this as an opportunity to cook healthier meals, not all participants were able to do so, especially those most reliant on food support and those with limited cooking facilities.

Many participants described 'trading down' on the quality of their meals and resorting to 'basic meals', such as beans on toast, jacket (baked) potatoes, eggs, and chips. For Kathy, her new financial struggles and difficulty in physically accessing food caused her diet to plummet considerably, as she explained how she could often not even afford butter or cheese to have on her potatoes.

> I've been having soup and things like that. Jacket potato with cheese on. Sometimes you have no butter or cheese at home. You have to have a jacket [potato] with nothing on. I like chilli con carne. I haven't had that for ages – tin of chilli con carne. (Kathy, age 50-60)

Liam similarly explained how his diet has changed since COVID, as he was no longer able to afford the foods that he wanted and had to depend mainly on food support.

> I'm a meat and veg type of guy. I could have a Sunday dinner every single day if it were me. Probably three times a day. If it was just me, I'd eat meat and veg. I would just love that. I've mostly been having to go and buy a cheap box of eggs. Obviously, you get the beans and that from the Foodhall. You might end up getting a bag of potatoes and have chip, egg and beans and a bit of bloody bread or something. Yes, my diet has 100% totally changed. (Liam, age 40-50)

In addition to cuts made in regard to food, many participants found themselves needing to go without other services. Several participants cancelled their internet, which increased barriers to connecting with social networks, finding available community services, searching for work, and applying for benefit programmes. This coincided with the closure of many public libraries, city services, and job centres, which might have previously provided internet access and job and benefit support services.

Thus, we see that even though participants were resilient, adaptable, and capable individuals, expending significant time and effort and utilising a wide range of individual skills and strategies to meet their needs, their efforts were often not enough to stretch their resources. COVID-19 created additional challenges to coping with food insecurity and also diminished some of the options available as services closed.

Linda described how her individual options or obtaining food and managing the other expenses in her life were diminished during the pandemic. Although at one time Linda had been financially 'well-off' and did not worry about financial struggle, her split with her partner in the U.S., loss of employment, and increased living expenses left her without enough to manage on. Without a valid visa, her options for accessing national support options were further diminished.

> I have spent much of my life [in this country], I have spent well over £100,000 in this country when I had money - in businesses and in property and you know, all sorts of things. Now, I mean, this pandemic, I just want to just have a laugh. There's nothing that I can do. There's absolutely nothing. I couldn't return [to the U.S.]. My parents are dead. You know, not that they would help me. Anyway, but you know, what can you do? And I had left a bad relationship to come back here [to the UK]. (Linda, age 40-50)

Even though Linda used meticulous budgeting and meal planning strategies to stretch her income, this still wasn't enough for her to be able to afford both rent and food. This realisation and struggle, also felt by many other participants, led to feelings of shame and a lost sense of control over their lives. In these situations where participants found that they had no other options available to them in order to manage on their own, they felt 'forced' to reach out for external support.

5.2.2.2 External support options and barriers to accessing them

Thus, as personal resources were stretched and individual strategies were not sufficient to cope with increasing financial and health struggles, participants turned to external options for support. Most participants only reached out for external help once all other individual options had been exhausted. External support included personal networks (e.g., friends, neighbours, and family), government services, and local, community-based organisations. However, accessing these options also came with their own challenges.

Difficulties arose as participants struggled to navigate government, council, and community-based support schemes to find available help, whilst many traditional means of support (such as visiting community centres and churches) were closed. At the same time, many of the social support mechanisms participants previously relied on were also disappearing or being exhausted, if they had them at all. Long-standing issues of social isolation came to the forefront, as many participants felt they did not have anyone they could talk to about their problems or rely on. Others were unable to see friends or family because of the need to isolate, which weighed heavily on them. Finally, many did not want to call upon their friends and family members during this time, realising that others were also struggling and not wanting to be an additional burden.

5.2.2.2.1 Personal networks

In times of need, participants often relied on their own personal support networks, albeit as a last resort. Friends, family, and neighbours provided options for splitting household costs, sharing services, and were sometimes able to help out with money or food. Indeed, all participants who had been in a situation during COVID where they did not have food in the house and could not afford or obtain more stated that they would reach out to friends or family for help in this case. Social networks were also invaluable sources of information for learning about other available support options and for providing emotional support during this stressful time.

However, most participants did not want to reach out for help from personal networks, either because participants realised others were also struggling during this time or because they were too embarrassed or ashamed to do so. Although many participants were reluctant to ask for help, most did identify people that they felt they could rely on when in need.

Linda provides an excellent example of how engaging with personal networks helped her during a time of need. Even though Linda did not have many close friends or any family in the UK, as a fairly newly arrived U.S. citizen, she was able to expand her social network during the pandemic to help her through the challenges she faced. Linda cooked for her neighbours in exchange for using their internet so that she could work on her visa application. She was thus able to use her skills of batching cooking and meal planning gained when cooking for her large family, to exchange for a service she could not afford (internet). She also befriended government employees to help her with her visa application, which had expired several times due to unresponsive government services.

> Everything's been closed. So, all I can do is email people. And what I've done is I've befriended people. I befriended someone in immigration. All I had was a broken tablet, so I could barely see. I befriended someone at my bank who's been sending information [for me]. Then, because the stuff [electronic documents] was so expired, what we did - there are 159,000 people behind on scanning information for, so this means that there's that many of us behind - is I had my stuff forwarded to the person that I made friends with. And then he made friends with the guy dealing with it. But what if, what if I was from a third world country? What if I didn't know that? What if I couldn't, you know, speak English? What if I'm on the street starving? (Linda, age 40-50)

In this statement, Linda shows the importance of making and having social connections to get the support one needs in times of crisis. However, she also acknowledges the privilege she has in being able to do so, recognising that that those who aren't able to make these connections are often left behind.

Other participants also consistently relied on help from others during this time. For example, Randy, who was self-isolating because of illness, relied on his neighbours to pick up food for him and his family from the shop before he found out about Foodhall's service. Additionally, older participants who had adult children, such as Paul, Caroline, and Lilly, all confirmed that they relied on help from their children during lockdown while they were self-isolating. Paul and Lilly spoke of how their children brought them food, while Caroline provided an example of how her kids raced her to the hospital when she was having health problems.

> I had a health scare a fortnight ago and they [my sons] were both there and got me to the hospital and everything, so they are there for me. They've always been there for

> me, you know what I mean? It made it easier that they were here, if you know what I mean, because I got to the hospital quicker. (Caroline, age 70-80)

We therefore see the importance of having personal networks in place as a source of both immediate and consistent support. On the other hand, the lack of personal networks thus diminished the options available for coping with food insecurity. An example is William, an older man who lived alone and felt extremely socially isolated. In describing his isolation, William explained, "I've got no one, love, absolutely no one." He did not feel as though he had people he could depend upon, although he spoke to both of his children regularly. He outlined that most of his friends, as well as his son, no longer lived in Sheffield; although his daughter lived nearby, he explained that she had her own struggles in trying to care for her son who had a disability.

William's social isolation thus narrowed the number of people available to him for support. This was complicated by the realisation that other people in his life, such as his daughter, were also struggling and stretched or 'had their own problems.'

Thus, an additional barrier to utilising personal networks for support was the fact that so many people were struggling during this time. Participants often did not want to reach out to others for fear of being an additional "burden." For example, Michael (age 40-50) who had been struggling to find work for a long time, mentions that he feels he should not be using his parents for support anymore, explaining, "My mom and dad help me, but they're getting older so I can't rely on them for this." Additionally, Martin, who receives Jobseekers' Allowance, expressed that while he knows he can rely on his friends for help, he doesn't like to ask.

> It's very hard because I don't like to bother them [my friends] or borrow owt off them because they're struggling as well, you see. They've always turned around and said, 'If you're struggling, Martin, we'll help you out as much as we can.' But I don't like asking. (Martin, age 50-60)

We see how Michael and Martin have both recognised that the other people in their lives were also stretched or struggling and did not want to add to their problems.

This is also seen in the case of a younger participant, Emily (age 30-40), who was struggling to find work before COVID and was financially reliant on benefits at the time of the interview. Similar to William, Emily felt extremely isolated both before and during the pandemic, describing that she felt as though she had no one in her life that she could truly depend on. Her isolation thus reduced her options for finding support from personal networks. Although Emily did rely on her mother for help from time to time, she also acknowledged that she cannot depend on her mother for this, as "sometimes she can't even help me." Emily thus describes the seemingly helpless situation she has to find herself in before reluctantly asking her mother for financial help.

> I don't like asking her because I don't want to depend on my mom. I have to because I need it, because I don't have anything else. I can't think of anything else to do, really. (Emily, age 30-40)

We see that although Emily desires autonomy and independence, she feels trapped in a situation where she must rely on others. This idea of reliance, especially when there was no option for reciprocity, was often framed as 'wrong' and not something that should be done.

Gerald (age 30-40), who wrestles with an ongoing gambling addiction as well as severe anxiety, expressed this as well, saying, “I've relied on a lot of people, probably too many times”, showing that he believes that relying on others is not something he should be doing. This framing shows the neoliberal-ableist rhetoric that pervaded participants stories and experiences throughout this study, where autonomy was viewed as the ideal and asking for help was just wrong.

Thus, we see that having personal networks expanded the options available to participants to cope with food insecurity, while social isolation narrowed these options by having fewer people available for support. Collectively, these stories evidence how the resources available to achieve food security were diminished as social networks were stretched and exhausted, with participants generally not wanting to seek help from others and feeling like they should not be. Engaging with personal networks for support was easier for participants to do when there was a reciprocal exchange taking place, as with Linda cooking for her neighbours to use their internet. This was in contrast to asking for a loan from a friend or family member, which was often seen as an absolute last resort. As participants did not want to add to the problems of their friends and families at this time, they turned to other external support options.

5.2.2.2.2 State support

The UK national government provided various support options aimed to combat the direct physical and financial barriers to accessing food during the pandemic. These included benefits programmes for financial support, also available before the pandemic, as well as free deliveries of food to those shielding (people identified as ‘clinically extremely vulnerable’).

However, finding the programmes that people were eligible for and applying for them was difficult with support centres closed, government services stretched, and limited internet and phone access. We see that many people did not receive the support they were eligible for, thus ‘falling through the cracks’, and some were even negatively affected by the inability of the government to update existing policies and programmes.

5.2.2.2.2.1 Financial support through benefits programmes

Benefits programmes were one option to find financial support during this time, especially for those who were now not working due to the pandemic. Eleven participants in this study were relying on some sort of government benefit programme for financial support during the first UK COVID lockdown (not including state pensions). One of the most common benefits programmes nationally for people with low or no income is Universal Credit (Department for Work and Pensions, 2020), and this was also utilised by many participants in this study. However, many issues with Universal Credit payments existed during COVID, which increased financial struggle for participants during a time of crisis.

Universal Credit payments (as well as other benefit payments) were often insufficient to cover participants’ expenses, and by the end of the payment cycle, many were left without enough money for food. The insufficiency of payments was further compounded by the deduction of advance payment debts, which was not suspended during COVID.

The financial burden of these debts, amid an already insufficient and stretched income, was outlined by several participants. Diane explained how the amount she received from Universal Credit wasn’t enough to cover all her expenses, saying, “you don’t get enough to manage on." Diane received monthly benefits payments and found that by the end of the third week, she often didn't have any money left for food.

> I'm on Universal Credit. It's always been a matter of trying to juggle my bills with buying food as well on the amount that they give me. I do have a lot of debt that they take out as well. They take £90 a fortnight for my loans. (Diane, age 40-50)

Thus, we see how the inability of the Government to update existing policies on benefit payments that would have helped people during crisis, such as pausing debt repayments, left many people struggling without enough money for rent, utilities, and food. These issues were largely a result of the Universal Credit system being built in such a way that it did not provide flexibility during times of crisis; indeed, although some debt repayments were able to be suspended during lockdown, debts taken for advance payments could not be paused (Work and Pensions Committee, 2020a).

For those not already on benefits before lockdown, or for those who had never had to utilise government benefits before, there were further struggles in being able to actually find and apply for the programmes for which one was eligible. As demand for government support surged during COVID-19, receiving help to access government-provided services slowed as these services became overburdened and workers were sent home to work, furloughed, or off work because of illness. In addition, many local drop-in centres, which could provide support for benefits applications or access to other government programmes, were closed or extremely stretched.

The delay and struggle this caused is shown poignantly in Linda's story. Her situation was complicated even further as she was undergoing additional challenges in her UK residency status as an American citizen. Linda's visa had expired during lockdown, and she was unable to travel back to the U.S. due to combined issues of illness and financial struggle. She was thus ineligible for the COVID-19 financial support payments made by the U.S. Government, while she also could not apply for benefits in the UK without a valid visa. Linda described spending eight hours a day for months calling different government departments and Citizen's Advice, trying to secure help for her visa, unemployment benefits, and food support. Often, she was directed to different departments and then re-directed back to people she had called previously in circles, finding it difficult to talk to someone who could actually advise her. She approached this stressful situation methodologically and strategically, befriending people in different departments so they would help her and re-submitting her application every five weeks like clockwork, as it kept expiring before it was assessed by anyone.

> I've literally fought 40 to 80 hours a week contacting people until I finally got somewhere, and they took so long to process this stuff, all my electronic documents expired. I have been filing since the shutdown and what I've done is diligently, every five weeks, they shut me down. I do it again. And then I do it again. And so also now in this country, for women that are survivors of domestic violence, there's a programme that will help them out. But how would a person even know about that? I did not get that. (Linda, age 40-50)

Linda's situation shows how the delay in processing applications for government support left people dangling during a time of crisis. It also shows the difficulty in finding out about the programmes actually available to people, as Linda mentioned the support for survivors of domestic violence, which she was eligible for but never knew about previously.

Like other participants, Linda's challenge in accessing state support was further complicated by the difficulty of securing an internet connection. She could no longer afford internet, while at the same time, internet services previously provided by local libraries and community organisations were now unavailable due to closures. Linda explained her experience "squatting outside of my library to try to catch a beam of data" when searching for government support programmes online.

After a gruelling five months, Linda finally confirmed that she had received an appointment to renew her visa in the UK without having to travel back to the U.S, and she was even able to get the visa fee waived. This provided her with the opportunity to find employment and apply for benefits. However, we see that this was largely 'too little, too late' for Linda; although the Government did finally update visa support procedures, Linda was already two months behind on rent and had been struggling to afford food for five months by the time this was enacted.

5.2.2.2.2.2 Food deliveries

In addition to the difficulties in finding, applying for, and receiving urgent financial help, those with physical barriers to accessing food also found difficulty in accessing the government programme available to receive food deliveries. People who were shielding (unable to leave their homes) during the pandemic were entitled to priority delivery slots from supermarkets; however, it soon became clear that many could not access these, as they filled too quickly or were not available in all areas. Thus, as the national food supply chain proved to be inflexible and the market response to deliver food to those who needed it was not adequate, the UK national government stepped in and began delivering food parcels directly to people's homes.

During the first UK lockdown, the Government provided weekly deliveries of food and essentials to the homes of those who were shielding. Although the Government provided a far-reaching and critical service, which should be praised, such a large-scale, nationalised service undoubtedly missed people out. If one was not characterised as 'clinically extremely vulnerable' based on specific health conditions, then they were not eligible to receive government-delivered food parcels. However, there were many at the time who were vulnerable to the virus and who therefore decided to shield even if not specifically told to do so by the UK Government; this was the case for three study participants (Table 61). These people thus needed help receiving food, but were not included within the Government's food delivery response.

In addition, even those who were eligible for government food parcels still sometimes faced challenges in accessing these. This is seen with Paul's story. As an older man suffering from chronic obstructive pulmonary disease (COPD), Paul was supposed to be receiving government food parcels while he was shielding. He explains how he received government-delivered food parcels for only three weeks before deliveries stopped with no explanation, although these parcels were still being delivered to other houses nearby.

> I phoned [to receive government food parcels] on the Monday, for argument's sake. Three weeks later, I kept getting three parcels. After that, I got nothing, and it took me

> four weeks to carry on trying to find out why I'd been stopped to no joy whatsoever, until eventually, I got a text message saying that they were no longer doing it. That'd be around seven/eight weeks into it. Yet, every Wednesday, the same lorry pulls up on our road and delivers to the three [neighbours'] houses. (Paul, age 60-70)

Despite weeks of calling both government and council services to find help, the issue was never resolved, and Paul had to turn to community support for food parcels instead. Thus, we see that where business and government support proved to be inadequate, inflexible, slow, and often inaccessible, participants turned to local and community services, like Foodhall, for help.

5.2.2.2.3 Community support

During COVID-19, community-based support efforts rose to fill the gaps left by state and market responses to the crisis. Many existing community organisations adapted their services to fit the local needs that arose during the pandemic; this flexibility in a time of crisis, which could not be achieved by many national services, was essential. Emergency food efforts like Foodhall's were instrumental in 'catching' those who were struggling to access or waiting to receive state support and other conventional food aid options, such as food banks that required referrals. Other volunteer groups sprouted up across the country, such as the National Health Service volunteers and neighbourhood-based mutual aid groups, which brought food, medical prescriptions, and other essentials to people who were unable to leave their homes. These services were thus more accessible to a wider range of people as they often did not have strict (or any) eligibility requirements; Foodhall, for example, did not require any proof of need to receive food parcels. The smaller scope of these more informal and community-based responses allowed them to provide more personalised and fast-acting services, responding to provide almost-immediate relief to people.

Paul's story shows the importance of having fast-acting community efforts. After the Government had stopped delivering him food parcels with no explanation, Paul soon came to rely on Foodhall for deliveries of food while he was shielding.

> I started phoning you [Foodhall] up once every 10 days, something like that, and I've never had any dissatisfaction. It's always been smashing. No problem at all. Basically, I went from one phone number from the Government, which was useless, to you [Foodhall], which was all right, 100%. (Paul, age 60-70)

We thus see how the inflexibility and slow response time of the Government left Paul unable to leave his home and without any food – even though he was indeed eligible for food parcel deliveries. Eventually, the Sheffield City Council recommended Paul to Foodhall, which he found to be a more reliable and accessible service. It becomes clear that community efforts like Foodhall's were instrumental in providing a rapid response to get food to people at the time they needed it, especially as people like Paul 'fell through the cracks' of state support options.

Thus, another important characteristic of community-based efforts was their accessibility. As people struggled to navigate government bureaucracy, having easily accessible local services was crucial. The Sheffield City Council also operated an emergency food parcel delivery service during this time, which consisted of a one-time only delivery of a week's supply of food that was supposed to be delivered in 24-48 hours. However, this service was designed for one-time support for those who were facing a COVID-related crisis, and thus those who were experiencing non-COVID related financial crisis, or who needed consistent support due to self-isolation over the entirety of lockdown, were often signposted to other organisations like Foodhall. Additionally, many who came to use Foodhall's service reported extended wait times and difficulty in accessing the council's support. Because of these issues, Foodhall provided a service which did not require any proof of need to receive a food parcel, as the project was aiming to catch those who were perhaps ineligible for other services or to provide 'meantime' support while people were waiting to receive the services for which they were eligible.

Michael, for example, expressed how it was much easier to access Foodhall's service than it had been for him to access food banks in the past, which required referrals. He expressed the challenges in first obtaining an appointment with a doctor to receive a referral to the food bank, and then the further challenge in transporting the food back to his home, when he was already struggling financially.

> Usually, it's difficult to get an appointment with them [the doctor]. It's trouble to get their referrals [to the food bank] in the first place and then transport was an issue. Usually, they're far away. They're in the same postcode, but they end up being miles away since the postcode's so round. And we're all the way on the other side of town, so it's a bit difficult. (Michael, age 40-50)

Michael thus reached out to Foodhall for help during the pandemic, as this service did not require referrals and delivered directly to his home. This accessibility of this service meant that Michael was able to receive food support at the time that he needed it, without having to wait for a food bank referral.

However, the informality of organisations like Foodhall also created challenges in accessing them. Because of their community-based nature, it was often difficult to find out about them in the first place; those without internet and phone connections faced further barriers. The fact that many of these efforts relied on donations also meant that most could only offer a narrow set of options for food and other essentials, creating further barriers in participants' ability to utilise the food they received.

5.2.2.2.3.1 Barriers to accessing support

Many people in the UK were now having to access food support for the first time in their lives. This meant they were new to looking for and finding out about what support options were available to them in their local area. For others who were using food support before COVID, the closure of some services and creation of new services created confusion about what help was actually available. Indeed, many traditional spaces where people might have gone prior to COVID for a hot meal or for advice on available support, such as community centres and churches, were now closed with lockdown.

The difficulty in finding out about available support meant that many people were left struggling for some time before actually receiving the help they needed; some may have never gotten this support at all. Martin provides a clear account of the difficulty in finding out about available support options. Although Martin had been struggling financially before COVID, he had never known about food banks until crisis struck during the pandemic.

> I was struggling [before COVID], but nobody told me about food banks then. This is the first time I've known about it. I didn't even know. (Martin, age 50-60)

The fact that many did not know about food banks or other food support options at the start of the pandemic shows how difficult it was to find out about these services in the first place; with additional isolation requirements imposed by lockdown restrictions, this only became harder.

The inability to leave the home complicated new or existing financial struggles, leaving people scrambling to find new ways to obtain food, as in Randy's case. Randy had never had to use any type of food support before COVID. At the start of lockdown, he did not know about food banks or Foodhall and was living off "basic rations" provided by his neighbours while he and his family were shielding. In some cases, Randy even skipped meals to ensure that his wife and daughter had one.

> I used to make sure my daughter and me wife got a meal. Other than that, I wasn't bothered about what I had because I didn't know about the food parcels what you could get. Now I know about that. We know that if I do run short, I can always nip up to the food bank and they would help us out, because they said they're going to be running all the time anyway, for people who are on low income. (Randy, age 50-60)

Once Randy found out about his local options for food support, this eased the stress of making sure that his family was fed, and Randy no longer had to skip meals. We thus see how critical it was to have these fast-acting community services available to 'catch' people in times of crisis. However, we also see that many still faced difficulties in finding out about and accessing local services, as Randy spent some time forgoing food before he received the support he needed.

Social networks were participants' main source of information about support options. In this study, ten out of the fourteen participants reported that they learned about Foodhall through word of mouth – either from friends, family, neighbours, or support workers. This showed the importance of having and maintaining social connections. However, isolation requirements and the closure of many community centres during COVID created an additional barrier to people communicating and sharing information. Because of this, the most isolated individuals were likely also the ones left behind.

Those without an internet or phone connection faced an additional barrier, as these were often required to find out about or utilise available services at the time. For example, Foodhall required people to call a helpline each time they needed a food parcel; many mutual aid groups organised through social media channels, like Facebook (Ntontis *et al.*, 2022). With public libraries and other places that might have provided internet now closed, this created a further barrier to finding out about or using available services.

Recognising these barriers, the Sheffield City Council began calling people directly to convey information about available support options. Linda explained how the council's ability to recognise and respond to this local need was a big help; however, this took about five months from the start of the pandemic to happen, meaning that many were likely left struggling for a long time before receiving any information.

> The people that helped are the ones that adapted. Just like the council started calling people because they realized a lot of us couldn't get online. I couldn't get online. I was squatting outside of my library to try to catch a beam of data, and I couldn't log in, all I could do was strictly hit send… And plus, with people who are older, you know, they're not gonna do that, and nobody was there to man [the] phones. So I'm glad some people from the council started calling us. And that's what helped. But this has taken – what? April, May, June, July, August – it's 5 months. (Linda, age 40-50)

In some cases, even having enough phone credit to call support services proved to be a further barrier for those struggling the most financially. These barriers are presented in Kathy's story. During the interview, Kathy explained how panicked she was when she ran out of money, and thus was unable to purchase phone credit to call Foodhall for an emergency food delivery. The trauma and anxiety this caused Kathy was palpable throughout the interview, as she kept referring back to this situation and repeatedly asked for confirmation about her upcoming food delivery.

> The other week when my phone got cut off, I couldn't get in touch with Foodhall. Then you got shut down. I was panicking. (Kathy, age 50-60)

Those who struggled the most financially had their options for support further diminished by the inability to access internet or phone services. For Kathy, this issue was compounded by Foodhall's temporary closure in early August 2020, when the organisation was moving buildings. Although all food parcel recipients were notified of this and were signposted to other organisations, this still left some people, like Kathy, panicking about where she would get food in the meantime. We thus see that the grassroots and informal nature of these community services, which allowed them to be more accessible and adaptable in times of crisis, was set against a need for consistency.

5.2.2.2.3.2 Barriers in utilising food

Once participants did find out about and access community food support, there were still barriers related to using the food that was given to them. Although all participants expressed extreme gratitude for Foodhall's service and a general satisfaction with the content of the food parcels, there were some cases in which foods could not be utilised. This was also an issue in government food parcels (McNeill, Dowler and Shields, 2022), and indeed was one that Foodhall tried to address by operating a more personalised service, where volunteers called each parcel recipient individually to curate a parcel in line with their dietary restrictions and needs. However, the fact that recipients could not choose every item in the food parcel undoubtedly meant that some foods were perhaps unable to be utilised, often spurred by unfamiliarity with certain items. This created an additional barrier to utilising food and thus achieving food security. Liam provides one example, as he often struggled to cook complete meals out of the combination of items he received in the food parcels.

> A lot of the stuff that I've got still in the cupboard is from the Foodhall. It seems as though every time you get one, you get a hell of a lot of the same things that you've got, so you might end up with bleeding nine tins of tomatoes. I'm not going to be making spag bol [spaghetti bolognese] every day, when I ain't got no mince to fit in the freezer...If you do any parcels, remember my name and start putting stuff in there that go together [laughs]. (Liam, age 40-50)

Similar to Liam’s build-up of tinned tomatoes, Martin also mentioned having a build-up of kidney beans in his cupboard, saying, “I've got about six tins of them, but I've not tasted them so I don't know what they're like.” We see that while both Martin and Liam were struggling with food insecurity, the inability to choose specific food items created a further barrier to utilising the food that they did have.

The unfamiliarity with vegetarian meals was also expressed by several participants. As Foodhall aimed to provide cooked meals that met the dietary needs of the majority, volunteers normally cooked vegetarian meals. In some cases, this may have been the first time that participants were eating certain vegetarian meals, so this may not have been something that they were comfortable with. William, for example, mentioned how he thought that Foodhall “shouldn’t be forcing it [vegetarian meals] on people when they don’t want it.” From these conversations with participants, it became obvious that it was important for food distribution efforts to be more cognisant of the typical foods that people were comfortable with cooking or eating, relevant to cultural practices.

It was also important to note the cooking facilities that people had. Again, this is something that Foodhall volunteers asked when calling parcel recipients, but in some cases it became clear that more specific details should have been gathered. For example, while Liam mentioned that he did own a freezer, volunteers perhaps did not realise that it was not large enough to accommodate all the frozen meals that Foodhall was delivering.

> You get pots of stuff, like soups or whatever, frozen, and the fact that I can't fit them in the freezer anyway, and then because they've been left downstairs and the nutters downstairs haven't rang me up to say that you've got a parcel. Then I get the bag upstairs and they're all melted and dripping inside the bag and everything. (Liam, age 40-50)

We thus see that including foods that could not be stored or cooked with available facilities, or which were unfamiliar and didn’t seem to ‘go with’ other foods in the parcels, may have influenced people’s ability to utilise food. In most cases, Foodhall addressed these challenges by providing a more personalised service that did allow people to request or exclude specific items from their parcels, something that was not possible in the Government food parcel distribution effort. However, it is possible that not all food parcel recipients were aware that they could ask for certain foods to be excluded in the parcels, or perhaps they did not feel comfortable making these requests. In these cases, participants may have been unable to utilise specific food items, which created another barrier to achieving food security.

This could potentially be alleviated by offering services that allow people to choose their own foods. Indeed, Emily expressed that she would have liked to have been able to have more choice over the foods in her parcel. However, for organisations like Foodhall with limited volunteers, a trade-off then erupts between serving a greater number of people or serving less people in a deeper capacity.

Throughout this entire section, we see that even with the struggles that all participants were facing during COVID, participants were resilient and organised in the ways they coped with the new financial and health challenges they faced. However, individual strategies were not enough to cope with the challenge of food insecurity atop additional struggle induced by the pandemic. As participants’ personal networks were also stretched, they turned to other options of external support. In many cases, government support proved to be unreliable, and thus participants utilised local, community-organised support efforts like Foodhall for help. These informal support efforts came with their own barriers, as finding out about them in the first place was often a challenge.

Even as participants did find available support options, they then faced additional barriers in asking for and accepting help. Mental barriers related to self-stigmatisation and grounded in a neoliberal-ableist ideal came to the forefront, as having to rely on others for food shifted one's sense of personal identity. This left people feeling a lack of agency and autonomy as they lost control over one of their most basic needs.

5.2.3 Stressed: The mental load

The crushing stress of finding ways to manage day-to-day during a pandemic, along with the complex and often heart-breaking calculations made between different needs, took an intense mental toll on participants. Despite the fact that so many others were also struggling during this time and much of what was happening – in the midst of a global pandemic – was out of their control, participants came to individualise their burden, characteristic of a neoliberal rhetoric that favours personal responsibility. This led to feelings of shame and failure about their situation, which was associated with a loss of autonomy and their identity as someone who could provide for themselves. These changing internal perceptions led to self-stigmatisation, as most participants believed that they should be able to manage on their own, but just couldn't.

In some cases, this shame and stigma resulted in participants isolating themselves from others, which separated people at a time when support was needed most. Increased stress, feelings of stigma, and social isolation also impacted mental health, creating a further barrier to coping with food insecurity as this reduced the ability to plan ahead and prevented people from reaching out for help at a time when recovery could have been easier to achieve. We see how this cruel and reinforcing cycle between mental health and food insecurity plays out, as one compounds the other in ways that further diminish the critical resources needed to achieve food security.

5.2.3.1 Changing personal identity

As participants came to rely on others for food, some for the first time in their lives, this led to them to question their own identities as independent, active, and contributing members of society. Losing the ability to provide for themselves or their families, or make decisions about one of their most basic needs, stripped away participants' sense of autonomy. The need to rely on others was often viewed as 'wrong', as participants felt that they should be able to be completely independent, even during a global pandemic. This shows how participants embodied the neoliberal-ableist desirable personhood as an independent, autonomous, and self-contained individual; when they tried but could not meet these rigid expectations, they began to feel 'less than.' Reaching out for external support was thus usually seen as a last resort, only done once participants felt they had no other options left. Relying on others led to feelings of embarrassment, shame, and a lost sense of pride in oneself. In some cases, the lack of agency and control left participants feeling entirely helpless.

Randy provides an example of how personal identities shifted during the pandemic, as external circumstances became internalised as personal failures and burdens. As a father and full-time carer for his wife, Randy found himself using food support for the first time in his life during the COVID lockdown. As his family was shielding due to health conditions in the family, Randy was unable to physically obtain food and was relying on help from neighbours until he found out about Foodhall's service. He describes how having to rely on others for food caused him to question his own identity as a provider for his family.

> I don't know to explain it. I felt ashamed of using it [food support] because I've always been able to provide for my wife and family and that. I felt a bit ashamed for asking for the handout. I know there's a lot of people in the same situation. (Randy, age 50-60)

We see how the lack of options available to Randy to obtain food for his shielding family led to feelings of shame and a loss of self-reliance. Even though Randy recognised that there were many others in the same situation at the time, he still viewed his need to rely on food support as an individual and personal failure to his family – characteristic of the neoliberal loading of crisis onto the individual.

This idea is expressed time and time again, as many participants internalised and individualised their struggles, even though these struggles were largely spurred by a global pandemic that was out of anyone's control. Many participants reported feeling like they should not need to ask for help and that they should be able to "manage" or "get on" on their own, but just couldn't. The idea that reaching out for external support was 'wrong' shows how participants placed the widespread struggles of the pandemic onto their own shoulders – an individualism that pervades neoliberal rhetoric.

This individualisation of burden is also seen within in Diane's story. Just a few months prior to lockdown, Diane suffered through a highly traumatic experience as she left her home and abusive ex-partner. During this tumultuous time in her life, she questioned her own identity as a self-reliant woman. Diane worked for 24 years before her previous employer closed in 2019; her ex-partner, however, spent much of her money on drugs, causing "a lot of friction" between them. Now unemployed and on her own, she feels ashamed having to rely on Universal Credit benefit payments and on her daughter for help. She feels this mental load of 'not being able to manage on her own' very personally, despite the immense struggles she has overcome in leaving an abusive relationship. She explains how she tries to survive off of the Universal Credit payments she receives, but it just is not enough to get her through the month.

> People work for their own money. I should be able to manage on my own now. I'm 41 years old. You don't get enough to manage on. It's hard to explain. I think it's just a pride thing, probably. (Diane, age 40-50)

In Diane's case, the inadequacy of state support translated to feelings of personal inadequacy, causing Diane to question her own capabilities and self-worth. This is again characteristic of a neoliberal-ableist framing where people must rely on themselves to make things work.

This idea that everyone should be able to manage on their own thus created another barrier to achieving food security, as participants felt they should not be asking others for help. Like in Randy's case, we see that participants even felt ashamed in accessing government programmes or community services like Foodhall's, despite the fact that so many others also needed help during this time. Martin further evidences this, as he describes his first experience receiving food from a food bank.

> I thought, 'I know I shouldn't be doing this, really. I should be doing it myself, getting through,' but I thought, 'Well, I got to get some help from somewhere.' That's why I asked the food bank. (Martin, age 50-60)

Martin's words show the stigma that developed around receiving support, as he felt that this was not something he should be doing, but also felt like he had no other options. Caroline provides a further example. As an older woman, Caroline found herself supporting her two previously self-employed adult sons during the COVID lockdown. Even though Caroline was eligible for food support, as she received fixed income through a state pension, she still felt shame and embarrassment in having to rely on it.

> I used to go down to Tesco's [the supermarket] before the pandemic started, and I used to see the boxes for the food bank and the different charities that Tesco support. They used to say to me, 'You know you're eligible?' I used to go, 'What do you mean?' 'Well, you're a pensioner on a limited income, you're eligible.' I said, 'No, while I can survive on my pension, I will do so, and save stuff for other people.' Unfortunately, the way things have gone with this pandemic, I have been really struggling, as I'm sure have a lot of other people. I had to swallow my pride and do something. (Caroline, age 70-80)

Through Caroline's words, it is clear that she believes that she should only be using external support as a last resort – when she can no longer 'survive' otherwise. This is despite the fact that she indeed recognises that other people need help from time to time. The burden that she places on herself to be able to cope on her own, when she doesn't necessarily hold this view for others, such as her sons, shows how Caroline has so severely individualised her struggle; this has happened in such a way that the idea of herself receiving support is linked to a loss of independence. Caroline's situation also shows how the inflexible and slow state responses at the time placed the burden of caring for her adult sons onto her shoulders. The lack of sufficient support thus not only left people struggling financially, but also severely affected how people viewed themselves and their own sense of self-worth. This, coupled with an individualisation of struggle, created further mental barriers for people to access the help they needed.

The fact that many participants waited until the last possible moment before resorting to external support options meant many found themselves in extremely dire situations before receiving the help they needed. Michael highlights this as he explains how he waited until he was 'starving' until he finally reached out for food support.

> Well, it was a relief [to get food support] when I was starving. It's still not good. You want to be reliant on yourself, don't you really? (Michael, age 40-50)

Michael's words relay his desire to be autonomous and self-reliant – in such a way that this prevented him from seeking support until a time of extreme crisis. He further expresses his lack of available options as he says, "We just have to get used to it [the food we're given] really, or otherwise we starve." This helpless situation, now tied to having to rely on others for food, thus diminished Michael's sense of agency and control over his life. This is reflected in the ways that those most reliant on food support now lost the ability to make choices about one of their most primary needs.

The lack of available options and loss of control over food sometimes resulted in participants being placed in uncomfortable and dire situations. For example, Kathy explained how her complete lack of food at times placed her in situations where she felt forced to eat foods she could not stomach, such as pork. During one time when she did not have any food in the house, she described how her neighbour brought her a bacon sandwich. Throughout the interview, she often referred back to this instance to express her extreme guilt for not eating the bacon.

> I really regretted chucking that bacon. I cooked it, but I couldn't eat it. I felt right sick. He brought me some bread and all, so I ate bread. I shouldn't have wasted it. I felt guilty…The stupid thing is, I went downstairs to another bin area, and somebody had left half a loaf on the floor, and I took that home and made some toast. Then I felt really guilty. (Kathy, age 50-60)

Because Kathy was self-isolating and unable go out to get food, and often could not afford it, her options were already limited; when presented with food she could not stomach, she felt mentally stretched to a breaking point, where her only available option was eating out of the bin. This further diminished her sense of self-worth and autonomy. It becomes evident that the struggle of food insecurity takes a physical toll, such as seen with diminished quality of diet, as well as a mental toll, both of which affect one's capabilities to achieve food security in the future.

We thus see how losing control over something as simple as choosing one's own food translated to a sense of lost control over one's life. Participants felt an acute sense of shame and personal failure in their struggle, even though many others were struggling for similar reasons during this time. It becomes apparent that many of our participants felt like they should not be asking others for help, and that this was wrong; instead, they should be a completely autonomous individual. This embodied neoliberal-ableism narrowed the options available to people to cope with food insecurity and other life challenges, as it in some cases delayed or prevented participants from seeking help, or it further diminished their mental capabilities by creating feelings of stigma and lowered self-worth.

5.2.3.2 Stigma

Compounded feelings of shame and loss of independence led to self-stigmatisation, as participants came to view themselves as outsiders of normal society. In this study, we see that participants often drew a line between the 'norm' as the neoliberal-ableist ideal - someone who can be self-reliant - and the 'deviant' as someone who needs to continually accept help from others. Several participants internalised this stigma as someone who could not provide for themselves in such an intense way that they felt utterly helpless, socially excluded, and unable to change their situation or engage as a reciprocating member of society.

These feelings of stigma created additional barriers to accessing support. As seen in the previous section, it delayed people in reaching out for help, at times when their struggle may have been easier to cope with. In other cases, it completely prevented people from accessing certain services or accepting help that was offered, for fear of how others would perceive them.

The ways that participants embodied the idea that one should be able to be completely self-reliant were based on views held over a lifetime, as neoliberal rhetoric long pervaded societal norms and Western policies. Because of this, even though conditions had changed during COVID (i.e., many were now struggling), it was difficult, and in some cases impossible, for participants to shift how they viewed the 'norm' in their minds.

Caroline provides an example, showing how she has carried the view of having to be completely self-reliant throughout her life. Through the stories that Caroline shared, we see that the embarrassment and loss of pride that Caroline felt when using food support during COVID ran deeper, linked to a stigma around having to accept help from others and not being an able and autonomous person. This was detailed as Caroline spoke about the experience of having to sign up to unemployment benefits ("we used to call it dole money") when she younger and lost her job. Caroline explained how she had always worked throughout her life, which gave her a sense of self-worth and independence. Despite the fact that Caroline was entitled to unemployment benefits when she was laid off due to cutbacks, she felt like she should not be using this state support, even for situations out of her control.

> The only place you could get another job was through the dole. You had to go and sign on, so you could get another job. I used to feel really awkward when I got the money every week. I said to my dad one time, 'I feel terrible taking this money.' He said, 'You've paid your National Insurance. That was what that's for. It's to help you when you are out of work.' These days, obviously, being a pensioner, you have to get over your embarrassment, don't you? You have to swallow your pride and do what you have to do for your family. (Caroline, age 70-80)

In this story, we see how stigma created a mental barrier to accessing support. Caroline expresses a desire for self-reliance, which is set against 'doing what she has to do' to provide for her family. We see how this stigma that Caroline held around accepting help when she was younger has carried through to how she now felt having to receive help during COVID – again for a situation that was outside of her control. Caroline's story provides further evidence of how participants viewed external support options as a last resort, only to be undertaken when there are no options left. This further cements the stigma that needing help from others is deviant to the neoliberal-ableist norm and not what one should be doing.

Although Caroline's embarrassment might have delayed her in accessing support, in other cases, feelings of shame and stigma completely prevented people from accessing certain support options. William and Liam both described how embarrassed, ashamed, and humiliated they felt when they visited food banks, which prevented them from ever returning to these places again. The shame and fear that they had of 'being like the others there' created a barrier to accessing a food support option that was available to them.

Throughout their interviews, William and Liam expressed views which showed how they stigmatised others at the food bank. They referred to other people at the food bank as "fiends" who are "taking the piss", perceiving them to be taking more food and other items than needed. William describes this sentiment:

> People are donating clothes, shoes, whatever, things like that, and they're [other people at the food bank] coming in with their wheelbarrows more or less and just taking the lot, and nobody gets a chance, nobody gets lucky, with these people, you

> know what I mean? They're fiends, they're just fiends. They're nothing more than fiends. Whatever they're offering [at the food bank], they'll take it, and it really upset me to think that these people would be taking about, from a church… I said I'm not coming here no more, because it's just riling me up, you know. I just stopped going down. (William, age 60-70)

Liam similarly described feeling "so out of place" at the food bank as he recounted his experience.

> There were people there [at the food bank] that were just, I don't know. It were like it was just natural to them, and it's like, 'I've got five kids, can I have more of them? Can I have more of these?' It was as though they were just taking the piss. Sorry, it was just like they were taking the mick, and obviously when it got to my turn, I just said, 'anything will do, I'm on my own, I just need a little bit, just to get me through.' Like I said, they [the volunteers] were fine, but I just felt so out of place. I actually said, 'I'm never going back there,' and I've not been back there, and I wouldn't go back there. (Liam, age 40-50)

We see how William and Liam both drew a line between themselves and the others at the food bank. This led them both to refuse to go back to the food bank, for the shame of appearing as part of a group that they had stigmatised in their own minds. Liam feared becoming someone who viewed going to the food bank as "natural." As he had never had to use food support before, Liam was perhaps scared that going to the food bank would become a common occurrence in his life as well. He also held a fear of others in his life viewing him differently, perhaps in this same way that he regarded others at the food bank.

Indeed, in many cases the fear of being stigmatised by the people closest to them created a further barrier for participants in reaching out for help. Liam was so worried about how his family's perception of him might change if they found out about how he was struggling that he often tried to hide his situation, only reaching out to them as a last resort. He, like many other participants, spoke of not wanting to be a "burden" on his friends and family. Liam explained that he did not want his family "feeling sorry" for him or changing their perception of him as "a strong person that has never needed anything in his life."

> I just don't want people to feel sorry for me because I've never been that person. I've always been that upbeat sort of guy, and I hate to think that people feel sorry for me and are trying to offer me stuff. I've been offered to go to my nephew's and stuff for food and whatnot, and I just decline it because I just don't want to. I feel as though I'm putting on people, and I've never had to; I've never had to put on anybody. (Liam, age 40-50)

We see how Liam's fear of others viewing him differently created a direct barrier to him accepting help from his family. However, with the severe struggles our participants were experiencing, being able to connect to and accept support from personal networks was vital as a way to cope with newfound stress and struggle. Despite this, some participants tried to hide their situations from their loved ones, for fear of being stigmatised by those closest to them. Concealing struggle in this way can lead to further psychological distress from the fear of being 'discovered' and the inability to share their mental load with those closest to them (Pachankis, 2007).

This is seen poignantly in the cases of Gerald and Liam, who were both homeless for a time and hid this from their friends and families rather than asking for help. Liam explained how upset his sister was when she found out that he was living in his car after he had split with his partner at the start of lockdown. He then moved in with his sister, but still isolated himself from her within the household because of the embarrassment he had about his situation.

> Well, the first week [I moved in with my sister], I was still really upset and that. I just basically isolated myself in the bedroom most of the time. Whenever I came down and see them having food and everything it just-- I don't know. I felt strange…It just didn't feel right. I just felt exactly just like a burden. By the end of the three weeks, I started to get used to it. It did become good and I would have loved to have been able to stay there but to get my own place [emergency accommodation] you've got to be homeless, and that's what I ended up being. (Liam, age 40-50)

We thus see that the fear of his family viewing him differently made Liam isolate himself and prevented him from seeking help from his sister. Similarly, Gerald spoke about hiding his homelessness from his family in the past, rather than asking for help.

> I've been on the street. I've been homeless for a month last year [2019]. It was difficult. I didn't like it at all. It was winter, freezing cold, couldn't sleep. I said [to the people I know] 'I'm at a friend's, I'm all right', even though I was on the street. I kept saying that I was at a friend's, I didn't want them to worry. I just tell a lie to them. It's not a good thing. Don't get me wrong. I'm thinking more about them than my own situation. Luckily it was only a month. Thank God. I don't know how people do it for years. (Gerald, age 30-40)

Gerald later expressed in the interview that he knew that hiding his struggle from his family was not good for his mental health, and that he felt better when he finally talked more about his problems to others. Liam also expressed how, after a few weeks of being at his sister's, he actually felt better for being around his family. However, despite recognising that it was important to be around people and to share about their struggles, both Gerald and Liam still avoided asking for or accepting help from friends and family.

The stigma around asking for help was eased by the ability to engage in, or recognition of, reciprocal relationships, which provided an option for participants to feel more agency and control over their lives. Reciprocity circumvented the stigma of relying on others, as this related to a mutually beneficial exchange rather than accepting a 'handout.'

The ways that reciprocity facilitated exchange of support for participants is seen in the case of Liam. Although Liam was adamant that he did not want to ask anyone to lend him anything, he did acknowledge how being able to drive around his friends or family in exchange for petrol money helped him financially.

> I don't reach out to anybody to lend me anything. People, well not often, but a few times, usually at the end of the month, they'll ask me if I'll take them somewhere in the car. Then they'll chuck me petrol in so I don't have to worry about taking it out of my own money for my petrol, which helps and keeps me afloat for a little bit. Other than that, no, I would never borrow. (Liam, age 40-50)

We see that although Liam would refuse anyone lending him money (a one-way exchange), being able to offer a service in return provides him with the critical financial support he needed at the end of the month. However, we still see that Liam, like many others, maintained the view that to accept help would be ‘wrong’ and deviant to being the self-reliant ‘norm.’

Throughout these stories, we see how participants stigmatised themselves, embodying a spoiled social identity by perceiving themselves as socially different to a neoliberal-ableist norm. The way that participants embodied the crisis induced by a global pandemic as personal failures shows how they individualised a burden that many others were also feeling at the time. Indeed, participants were surprised to learn that others were also struggling, and that they were one out of a thousand people receiving food from Foodhall every week. Participants thus felt isolated and alone in their struggles, which was furthered by trying to hide their situations from their friends and family for fear of others in their lives also viewing them differently. This created further barriers to coping with stress and struggle, separating people at a time when support from others was so critical.

5.2.3.3 Social isolation

The feeling of isolation was widespread during COVID-19, as lockdown measures physically separated people from seeing their loved ones. Perhaps more than ever, the pandemic brought the idea of ‘social isolation’ to the forefront, with this word now becoming commonplace in everyday dialogue. Nine out of fourteen participants specifically reported that they felt socially isolated during the pandemic. However, for many participants, social isolation was an issue long before COVID – one that was amplified by lockdown measures and by the isolating experience of food insecurity.

Social isolation presented a barrier to coping with food insecurity because it narrowed the options available to people for financial, physical, and emotional support. Those who did not feel that they had others they could rely on felt an increased mental load, as they were unable to share their burden with others. Even for those who did have strong personal networks, the shame many felt about their struggles during this time made it difficult to share with others or ask for help. Increased stress and isolation, atop being physically separated from others during a global crisis, further affected participants’ mental health, making it more difficult to cope with other life challenges.

Some participants said outright that they did not have anyone in their lives they considered as a true or close friend that they could talk to about their problems, showing a lack of critical social resources needed to cope with crisis. As a younger woman who lives alone, Emily described her social isolation during COVID, which she said existed much before the lockdown. She expressed that she felt as though she did not have any true friends and that no one in her life, including her own mother, understood her.

> I've not really seen anybody [since COVID-19] to be honest. I've seen my mom a couple of times and my dad. That's it, really, but I don't – to be honest with you, I don't really have any friends. It's not as though I can just go out and see a friend. (Emily, age 30-40)

Although Emily mentioned that she does get help from her mom from time to time, she still felt as though she had no one she could truly depend on or talk to when needed. Social isolation thus presented a barrier for Emily to cope with her struggle. It reduced the options available to her to engage in reciprocal exchanges of support, thus limiting the opportunity to ‘share the burden’ or ‘lessen the load’ with others. This led to feelings of social exclusion and self-stigmatisation, seen in the way that Emily explains how the people in her life “don’t understand me.” With limited options for support available in her personal networks, Emily felt forced to turn to other options for food support. She described the helplessness and lack of agency and control she felt when visiting a food bank for the first time.

> Well, when I went there [to the food bank] I felt like I would be going to the food banks all my life and living off food banks all my life. That's what it felt like. (Emily, age 30-40)

Feeling unable to change her situation or engage in the reciprocal relationships that society is built upon left Emily feeling utterly helpless in her situation, seen in how she felt like she would be dependent on food banks for her whole life. We thus see how Emily’s severe social isolation was related to her feeling excluded from society.

Unfortunately, Emily’s situation was not unique; this trend of severe isolation existing before the pandemic was seen amongst other participants as well, young and old alike. William, a 68-year-old man who lives alone, described that all his closest friends “are either dead or they’ve moved out.” He talked at great lengths about how he felt that he can no longer depend on the friends that he does have, even just to show up to socialise when they said they would. Like Emily, William felt that it was always him reaching out to others, with no one ever calling him just to see how he was doing.

> I felt isolated way before the pandemic, sweetheart, like I said, fair-weather friends. The only time they really ring you up is if they want to know, ‘Have you got –? Can you do –?’ and I’m not like that. When I go and visit people, I go and visit just to see how they are, you know, I don’t go to see, ‘Have you got –? Can you do –?’ So this is why these fair-weather friends aren’t coming no more. I just give them a short bit. I just ask them, ‘What’s up? Can’t you just bloody call? You know, a cup of tea?’ (William, age 60-70)

Even though William does have friends and adult children that he talks to, he still feels isolated because of what he sees as a lack of true connection. This is seen in how William believes that his friends only call him when they need a favour.

Other participants also felt that their friends were inaccessible for social and emotional support. A commonly expressed sentiment was that their friends have “got a life of their own” or they were “busy with their own lives”, often voicing that jobs, partners, children, and health conditions took up the majority of other people’s time. Because of this, participants often did not want to reach out to their friends, feeling as though they would be a bother. Diane, for example, expressed:

> A lot of my friends now, a lot of them have got health conditions. A lot of them are still very much afraid to come out of the house because obviously this virus is still about. We've not met up to have a coffee or anything like that in months and months. Whereas we used to meet at least once a week before. I still talk to them on the phone, now and again, but it's just not the same because they're just busy with their own

> lives. I mean, they've got their children at home and things like that so they're always busy. (Diane, age 40-50)

Diane identifies how she feels certain things – like children – take up the majority of her friends' time, but also how COVID specifically created another barrier to her engaging with her friends through lockdown measures and the fear her friends had of getting the virus. Diane highlights how internet and phone connection were essential for staying in touch with friends and family during lockdown. Thus, for those who did not always have these, such as Kathy and Linda, an already extremely stressful time of crisis was worsened by not being able to stay in touch with loved ones.

Even for those that were able to stay in touch with others virtually, many expressed that this did not feel like true social interaction. The induced social isolation from lockdown measures in many cases made people realise just how important social interaction had been to their mental health.

> If they haven't got FaceTime, then I haven't been [in touch] – like Facebook and things like that, I could speak to them [my friends] on there, but that's it. I've not seen anybody physically, which – it sounds daft, but it's such a primal thing. Contact with somebody is just so weird. You just don't realise how bad it affects you until you're actually doing it. (Diane, age 40-50)

Diane expressed how the loss of physical contact was considered as a loss of something "primal" – indeed, a loss of basic human nature – which limited the depth of connection that one could have with someone else. Other participants also expressed how speaking to someone virtually presented a barrier in being able to truly connect and share about their struggles, which made it more difficult to cope with stress. Caroline talked about how her conversations with people during lockdown felt limited and insubstantial.

> I've got my front garden. I can go out there in the garden, or I can go out there, if it's really nice, and sit and have a coffee. People will go past and go, 'Hiya. You all right?' 'Yes, fine.' You still feel isolated because it's not really an atmosphere where you can sit and have a natter, if you understand what I mean. It's just generalised conversation. (Caroline, age 70-80)

Indeed, while Caroline has kept up with friends and family over the phone or across her garden gate, she feels that "It's not the same as going around and having a biscuit and a cup of coffee and a cigarette with them - it's not socialising." We therefore see how being able to share about struggle during COVID was limited by lockdown measures, as people found it difficult to engage in deeper conversations.

We also see that participants missed the 'organic' and 'natural' ways they used to interact with people – such as going over to a neighbour's house, meeting someone in the street, or even chatting with a shop employee. Lockdown limited these interactions but also made them 'wrong' or potentially harmful, as they presented a possibility of catching the virus. Caroline speaks about how she used to love having informal conversations with employees at McDonald's every weekend, and this is something she misses the most.

> I used to go to town every Saturday. It was my treat. Even if I didn't shop, I'd window shop or just go in places like Primark and have a look around. Then I'd go to

> McDonald's for a meal. That was my Saturday treat for myself away from my family. Away from everybody…Obviously, I haven't been able to do that for months. I'm missing that. Even though I didn't go with anybody, it was nice to get out of the area of where I live. I got to know the staff and some of the other people that used to go in on a Saturday. We got to be kind of regulars. Even if we didn't sort of chit and chat, it was, 'Hiya, hiya, you all right?' 'Hiya.' Do you know what I mean? They recognize people's faces and just acknowledged that they were there. I kind of miss it. (Caroline, age 70-80)

Through these stories, another common theme that comes to light is how people often associated these instances of social connection with food. Almost always, when participants expressed what they missed during lockdown, it was related to a social interaction centred around food. In previous quotes, both Diane and Caroline mentioned how they missed seeing their friends for a weekly coffee. Others expressed how they missed Sunday lunches out at a restaurant with family or a partner, or even just a simple chat with employees at a favourite restaurant or take-away. Paul describes this:

> I used to love going out on a Sunday for me Sunday dinner. We always used to go out on a Sunday because I would always say to my wife, 'We've been working all week' or 'We've been busy all week' or whatever we've been doing. I used to say to her on a Sunday, 'You don't cook – we go out for a Sunday dinner.' That's what we've been doing. We've always done that for years. That's another thing that I've missed, you know. A bit of socialising. Going for a drink or something like that, or going out for a meal. Even just going to Gregg's for a cup of coffee and sausage roll, something like that. (Paul, age 60-70)

From these stories, it becomes clear that food served as a connector for many. While lockdown restrictions prevented people from sharing meals with others, financial struggles and physical barriers also prevented people from accessing specific foods that held meaning in their lives. Participants thus lost the ability to connect with others around food and at the same time lost the ability to connect to their own foods, especially for those most reliant on food support. We see that even outside times of lockdown, the experience of being food insecure isolates people from each other as it often prevents people from engaging in social life (Runnels, Kristjansson and Calhoun, 2011).

5.2.3.4 Mental health

The newfound struggles and hardship faced during a global pandemic, coupled with a loss of autonomy, embodied stigma, increased social isolation, and other traumatic incidents that coincided with COVID, placed an intense mental load on participants. This crushing stress became even harder to cope with as people faced barriers to interacting with others, which severely affected mental health. COVID complicated relationships between people outside the household, creating anxiety toward others for fear of the virus, and inside the household, with increased stress and tension from re-arranged households and the need to stay inside. At the same time, many community and wellbeing centres, where people might normally interact with others undergoing similar life challenges or receive mental health support, closed with lockdown. For many, COVID was thus seen as a direct barrier to moving forward in life, leaving participants feeling helpless and trapped. With social networks and many other support options inaccessible, the issues associated with food insecurity, physical health, and mental health compounded on each other, creating a reinforcing cycle that became more and more difficult to escape from.

Physical and social isolation, financial hardship, loss of independence, and the overall lack of activity that resulted from lockdown created new mental health challenges for some participants and amplified existing mental health challenges for others. Five out of fourteen participants reported specific mental illnesses that they had before the pandemic, including depression, anxiety, and bipolar disorder, which became more difficult to cope with during lockdown. Other participants described feelings of increased frustration, stress, loneliness, and anxiety during lockdown.

> I feel like I've being stuck in the house. I get stir-crazy talking about other people. Some days I have chanced it and gone out with the dog, taking her for a walk myself when I felt really frustrated and stir-crazy. (Caroline, age 70-80)

For many, the stress of financial struggle and frustration of life in lockdown was furthered by feelings of aloneness. Diane described how the fact that she had to self-isolate over lockdown, during a time when she was also dealing with the traumatic split from her abusive ex-partner, impacted her anxiety and depression.

> I used to be a social butterfly. Now I'm not. I feel so isolated and so alone, nights, it's oh so hard. I started getting my anxiety and depression back again because I'm not seeing anybody. (Diane, age 40-50)

For Diane, the lockdown measures and resulting isolation that coincided with other traumatic incidents going on in her life made this trauma harder to cope with. Indeed, many people were experiencing trauma that coincided with the virus, such as newfound unemployment, loss of loved ones, or separation from partners and family. For some, other ongoing struggles, such as alcohol and gambling addictions in the case of Liam and Gerald, became more difficult to manage. For others, the pressure of lockdown even brought up feelings and reactions to past traumatic events, with William and Gerald both describing anxiety returning after separate attacks that happened to them many years ago.

> I was attacked about 30 years ago, and I still have nightmares about it. I was attacked for no reason at all, you know. I suffered broken ribs, arms, everything. I was just stuck in house for 8, maybe 10, weeks or more. I can’t quite remember, but it took about 3 years for my ribs to properly heal, so I’ve just avoided contact with people for like 30 years. (William, age 60-70)

We see how the trauma that William endured such a long time ago still affects him thirty years later and isolates him from those around him. Although William did later say in the interview that he now tries to socialise more with others, the ways he enjoyed doing this – such as by going to bike shows – was lost during COVID. We therefore see how past trauma or trauma that coincided with COVID became even more difficult to endure as participants were further isolated from friends and family. The loss of social interaction during COVID made people realise just how important social contact was to their mental health, even if they did not regard themselves as a “social butterfly” like Diane did.

This is seen in the case of Emily, who describes herself as an introvert. Although Emily, like William, felt extremely socially isolated both before and during COVID, she describes how she makes herself stay socially connected, if only for her own mental health.

> I've done it [kept in touch with people] just to make my mental health better for me, and my mental health generally. I'm thinking about myself. For example, if I don't talk to my mum for a bit, and then I'll think back, 'Well, I need to.' If I don't want to talk to my mum, I'll just text my mum anyway, because she's my mum. Even if I don't want to talk to my mom, I'll text her because I'm just thinking about me. (Emily, age 30-40)

Even though Emily acknowledges that sometimes she does not feel like speaking to others, she knows that it is important to stay in touch with her family.

However, COVID complicated social interactions between people. This happened as a result of lockdown measures, in the way that people were not allowed to just go over to a friend's house for a cup of coffee, but also in the way that the virus changed how people perceived each other. Someone walking down the street was no longer an opportunity for a chat, but now a potential threat – thus complicating one's relationship with their community. This change in perception was especially felt by those most vulnerable to the virus, such as the more elderly participants and those already living with chronic illnesses. Caroline, for example, expressed her frustration at how more people were using her local park during lockdown, which prevented her from being able to go there herself as she tried to self-isolate. Fear of the virus and lockdown measures thus limited the public spaces available to Caroline for fresh air and exercise. Although Caroline enjoyed socialising with others and having informal chats with people, she now had to view these informal interactions as potential threats to her health.

The fear of the virus and threat to health thus led to anxiety and worry for some participants, which undermined their sense of autonomy and left them feeling alone against the world. Paul, as an older man who was clinically extremely vulnerable to the virus, described his newfound fear and loss of independence as he had to shield and stay away from others during lockdown.

> I've not been going out at all. I'll be honest with you. I'm 66. Nothing's ever worried me in my life. I'm not even afraid about dying, but this has really stooped me. I'll be honest with you - I'm still apprehensive about going out. I can look where I'm going. It's other people that you've got to worry about. It's not just yourself. It's other people. There's a lot of them just don't give a care. They just come straight at you, walk straight towards you. (Paul, age 60-70)

This fear of others meant that what Paul took for granted previously—socialising and running errands for himself—created a situation where he now questioned his every journey and had to constantly work out how to avoid people. In this way, we see that COVID led to a loss of 'normal life' and complicated interactions and relationships between people.

COVID not only changed how people interacted outside the household, but in some cases led to complications inside the household as well. As household compositions were re-arranged prior to lockdown and many were now "stuck in the house" all day, this changed people's daily interactions with each other. For example, Caroline, who was isolating with her two adult sons during lockdown, described how they had begun “getting on each other’s nerves” from being inside the house all the time. Still, she generally enjoyed being around her sons and said that being able to have alone time in separate bedrooms helped – something that was likely not an option for all families.

For others, however, family situations were made more difficult by having people inside the house all day. In Liam’s case, the stress of the pandemic created additional tension in his relationship and also made it more difficult to manage his alcohol addiction. Eventually, this resulted in Liam being kicked out of his house by his partner of six years at the start of lockdown, after “going out on a silly bender.”

> If we could have gone out [of the house] and done stuff, maybe it would have been a little bit different. Just being locked in for all this amount of time, yes, I should imagine that caused what it caused [getting kicked out of the house]. That's why I'm here today. (Liam, age 40-50)

Now living in emergency accommodation, Liam describes how difficult it has been living away from his family, especially his children, whilst also dealing with his addiction and financial stress on top of that.

> The first day I got in here [emergency accommodation], I just couldn't believe where my life had gone, to be fair. The amount of days that I've just been in tears and not having a clue what to do, and just thinking shall I bloody end everything?...It's caused me to drink a few cans every night because I've stopped with - why I ended up here in the first place [drinking]. I haven't got bottles of vodka every single day. I've really reduced my alcohol intake, but I still like to have it in, in case I feel as though I need a bit. The money situation is just horrendous at the minute. I just struggle and struggle and get so uptight and stressed about it. (Liam, age 40-50)

Despite the challenging times Liam has had to endure, he explains that it is getting better day by day, and attributes much of these positives to his alcohol support worker. Many other participants also mentioned that the support workers they had been assigned for various disabilities or mental health challenges were an immense benefit to them during the pandemic; often, these support workers became a main source of social interaction. For Liam, who was embarrassed to talk about his situation to his friends and family, being able to share with someone less close to him, like his support worker, was extremely beneficial. Liam also expressed that it was important to be able to talk to a professional who understood the specific challenges that he was going through.

Thus, we see that the barriers to socialising during COVID meant a loss of ways to cope with mental health challenges. Like Liam, Gerald expressed a need to be able to share about mental health struggles and other specific life challenges with people who understood them. For the past three years, Gerald had been struggling from severe anxiety incited by a traumatic stabbing incident. Although Gerald expressed how important it was to not isolate himself from other people, it was often difficult for him to discuss his anxiety with his friends and family, because he felt they did not really understand it.

> I don't say that much about it [my anxiety], maybe on the phone I will talk to you about it. I might have mentioned it to a friend occasionally if I'm feeling really bad. No, there's nothing wrong with him. Somebody could look at me and say, ‘there's

> nothing wrong with him', but they can't tell inside. From the outside I look all right. My skin looks all right, et cetera. You must be fine. Fit as a fiddle. They don't realise what's going on inside…I try and hide it [my anxiety] a little bit. Because people don't always believe you. They think you're exaggerating it or something. One of my mates actually [didn't believe me] at first. He does know [about my anxiety], he didn't just always, because all my balance is off. He said, 'You're still walking, you can't be that bad, can it?' It got to me a bit actually because I thought if your own mates can't even believe you, it's quite worrying. (Gerald, age 30-40)

We see that Gerald experienced barriers to speaking about his mental health challenges because he was worried that people close to him would not believe him.

The way that Gerald approached this issue before COVID was by visiting community and wellbeing centres, where he could interact with counsellors and people who might have been going through similar life challenges. Indeed, prior to COVID, Gerald would visit a different community centre or lunch club every day, sometimes walking over an hour to reach one. Since he was struggling with his anxiety and was unemployed, going to these centres were his main source of daily activity and provided him with an opportunity to have a hot meal, socialise, learn new skills, get physical exercise, and volunteer. With these places closed during COVID, the resulting loss of activity severely affected Gerald's physical and mental health. Gerald expressed how difficult it was to fill the day, which led to trouble sleeping, heightened anxiety, and a resurgence of panic attacks that he had not had for some time.

> I haven't got a job. I'm unemployed. Obviously, because I'm on the sick, I'm not working, so I haven't got anything like that to look forward to. Getting up late is better around eleven o'clock, mid-day, something like that. I can cope with it then. Also, when I try to go sleep at ten, eleven o'clock on a night, I lay down and I get, it's like, a weird sensation. Like, my chest feels right heavy, and then l get nervous twitches with my mind. (Gerald, age 30-40)

With nowhere to go during the day, Gerald's mental health deteriorated. The boredom and lack of activity also led to trouble managing his lifelong gambling addiction. As Gerald began spending more time gambling online, this further affected his financial security and ability to afford food. We therefore see how Gerald's strategies to cope with food insecurity, financial struggle, and physical and mental health challenges were diminished with the closure of critical community spaces, which further affected his capability to be food secure.

Other participants also spoke about how the closure of critical public spaces created additional barriers to being able to cope with food insecurity and other life challenges. Some reported feeling "trapped" or like their life was "on hold", and thus felt unable to move forward. Diane and Kathy both provide examples of how the difficulty accessing mental health services during COVID impacted them.

Diane had recently left an abusive relationship just before lockdown. At the same time, the need to shield before and after a medical procedure meant that she was physically isolated from her friends and the other people who lived at the women's hostel where she was staying. She spoke of how she had tried to access therapy services, as a way to help her work through the traumatic experiences she was enduring, but that it had taken her three months from the onset of the lockdown to even get an appointment to discuss therapy options with a doctor.

Kathy, on the other hand, had been utilising mental health services before COVID, which had closed with lockdown. Also suffering from physical health issues, Kathy explains how frustrating it had been trying to cope with her bipolar disorder and depression without the necessary support.

> Since my dad died, I went a bit weird. I ended up getting a depression right bad…They've closed a lot of places down and they're not doing no mental illness things anymore for people. I don't know what's going off. Hopefully, I'll be able to get back on track. I don't know. It just pisses you off, doesn't it? (Kathy, age 50-60)

Kathy shows how critical the mental health support had been for her prior to COVID, and now expresses concerns about if she will be able to "get back on track" after these places closed. We see how the closures of critical services and community spaces during lockdown thus provided intense barriers to being able to cope with the combined issues of physical and mental health challenges and food insecurity.

Emily also described how she felt that COVID was a barrier to moving forward in her life. She spoke of how she could no longer go to the library to search for jobs, go to the gym to improve her physical health, or even just go out for a cup coffee, which left her feeling trapped.

> I think that's a good word to use, yes, trapped. It felt like I couldn't go out and do anything that I wanted to do, that I needed to do. Things that I needed to do to try and live my life. Try and get on with my life, to live my life, just simple things which I couldn't do…Normally, I would go to the library and search for jobs. I couldn't go to the library. I couldn't go to the gym because they closed all the gyms down. (Emily, age 30-40)

As life seemed to come to a standstill during lockdown, we see how this reduced the agency and the control participants felt they had over their everyday lives, which was compounded by financial struggles and having to rely on others for food. We thus see how food insecurity and mental health challenges influenced and compounded each other, and how the loss of social interaction and critical community services made existing struggles even harder to cope with.

Even with these heart-breaking stories of struggle, some participants were able to see the positives of life during the pandemic. Several used this time to improve their health, help others, or even to connect more with their family. Linda, for example, described how she used the time of lockdown to improve her physical health by going for walks, eating healthier, and doing yoga, and also improve her mental health, as she used meditation and reflection to work through emotional "baggage" from past relationships.

> I did a lot more meditating, did a lot more time in nature, being connected with that. Did a lot more introspective, lot of spiritual, getting grounded. So, I think I really cleared a lot of cobwebs, so to speak. I needed that. Because I was married when I was 19 and raising kids, so this is like, time to really break patterns, clear some issues out. (Linda, age 40-50)

We see that even with the immense struggles that participants were facing, many were still able to find ways to gain control back over their lives. Participants often expressed that the positives in their lives, or what helped most, were related to people – either friends, family, support workers, volunteers, or other community groups. Being able to give back and feel part of a community made people feel more active and connected to each other, at a time when in many others felt so far away. These stories underscore the importance of building community support networks to increase resilience and expand the resources available to cope with crisis.

5.2.4 Role of community

Throughout the previous section, we saw how social isolation and stigmatisation created additional barriers to asking for and receiving support. As participants came to view themselves as deviant to the neoliberal-ableist norm characterised by self-reliance and autonomy, this act of stigmatisation left them feeling like outsiders in society. Despite the rhetoric at the time of everyone 'being in it together' and needing to work together to 'save lives' (Department of Health and Social Care, 2021), participants individualised their struggles, leading to feelings of aloneness. What began to ease this isolation, stigma, and exclusion were the social interactions that made people feel less alone and part of a community.

This section thus explores the importance of social interaction and building community to provide opportunities for mutual aid. Knowing that 'you're not in it alone' – in direct opposition to the pervading neoliberal individualism – and being able to share about struggle eases the mental load of stress. Further, the opportunity to engage in reciprocal relationships lessens the stigma in asking for help as support becomes an exchange rather than an act of charity. Community organisations like Foodhall were important in connecting people, reducing stigma, and providing a platform for reciprocal exchange to occur. Community spaces are thus an essential resource to build social networks, which we found to be so important to cope with food insecurity and other life challenges.

5.2.4.1 Social interactions: sharing the load

Feelings of stigma and exclusion were lessened as people began to share with others about what they were going through and found they were not alone in their struggles. Social interaction thus became important for coping with the stress and mental health challenges that coincided with financial struggle, being food insecure, and living through a pandemic. This sentiment is captured by Gerald. Although he acknowledged that in past times of his life, he would try to hide his struggles from his loved ones, he explains how he has come to realise just how important it is to stay in contact with friends and family and talk about his situation.

> I think it's important to keep in touch with people. You don't want to get isolated. I've been down that road before, isolation, really depressing. I've had some people, if they're really depressed, I've had friends who just then don't talk to you for a while because they try to keep it to themselves. They don't want to talk to anyone. I think that's the worst thing you can do, personally… I did the same thing [years ago] and I just thought-- Since then, I've changed over a few years to start talking to people about it [my problems], which I found were better, actually. I learned from that, isolating myself from people when times are hard. I kept saying, 'I just don't want to talk anybody. Just leave me alone,' et cetera, but then I found it seems to be a burden on you. I've found that talking to people, it's really helped me. (Gerald, age 30-40)

Participants found that just being able to talk about what they were going through eased their mental load by being able to 'share the burden.' Sometimes, this was easier to do with people that they were less close to, especially for those who were worried about their friends and family perceiving them differently, like Liam. Whilst Liam felt he could not share his struggles with his sister or his son, he found that what helped him most was being able to talk things through with his alcohol support worker.

> At first it [seeing my kids] was awful because having to leave them and see them crying and not wanting me to go and arguing with the ex. It's got easier. It's got a lot easier. The main person who's really got me through it is my alcohol support worker, to be fair. She's clever and knows what she's talking about and she's dealt with a lot of this sort of stuff. (Liam, age 40-50)

Liam further explains how he finds it easier to accept help from his alcohol support worker and from community organisations like Foodhall than from his own friends and family.

> Because she [my support worker] is a professional lady, it's not as though I'm shy to speak to her and tell her my feelings and how it is. Whereas, if it was my sister or my son I'd feel as though they'd be feeling sorry for me. Trying to do stuff that I just don't want charity from them. I know the food bank is like a charity, that's just something I've got to do. I don't really know you. You're just trying to help people, regardless of who they are. It's just different when you're having to take help from your family when they've always known you as a strong person that has never needed anything in his life. (Liam, age 40-50)

This shows how critical it is to have holistic support available to address both mental health challenges and also ease logistical struggles, such as accessing food. It is also crucial for this support to be available in a way that overcomes barriers of stigma. Indeed, Liam expressed that he would never go back to a food bank because of how humiliating the experience was for him. However, he was able to accept much-needed food support from Foodhall because the food was delivered directly to his home, which removed the stigma around having to physically go in and ask for help from people.

Other participants also described how interactions with volunteers at different community organisations helped lessen stigma and made them feel more comfortable in asking for help. Randy, for example, spoke of the initial shame he felt using food support to feed himself and his family for the first time in his life. He found this lessened once he talked more with friendly volunteers and found out that many others were also struggling. This experience was echoed by Martin, who had also begun using food support for the first time during the pandemic.

> They're friendly people [the Foodhall volunteers]. When they deliver the food, they're helpful. If you need any help, you just ask them and they'll try and help you out and that. I felt I wasn't embarrassed. There's a lot of people out there now doing the same thing. (Martin, age 50-60)

We see that Martin's realisation that he was not the only one struggling at the time lessened his perceived stigma about using food support services. Recognising that the 'norm' had shifted and many people across the city were now relying on the help of others eased the shame and stigma around using food support. Randy and Martin were then able to view going to the food bank or getting food deliveries as positive experiences, chatting to volunteers or even making new friends there.

Knowing that others were also going through similar struggles perhaps lessened feelings of aloneness for Martin, by understanding that many people were also affected by the pandemic and his struggle was not just a personal failure. This was mediated by “friendly” volunteers who aimed to remove the stigma around receiving food support by letting people know it was okay to ask for help. This lessened the shame and embarrassment around receiving food from Foodhall or other organisations, as also expressed by Diane.

> I was a bit embarrassed, to be honest with you, the first time that I ever went [to the food bank], but like I said, they were so nice with me, you just lose that straight away. (Diane, age 40-50)

Diane spoke further about how her interactions with Foodhall and National Health Service (NHS) volunteers were some of the most positive moments for her during lockdown. Being able to interact and chat to volunteers was important for Diane, who was socially isolated during lockdown due to the need to shield before and after a medical procedure.

> I think the positive thing would be where I am [at the women’s hostel]. Also, interactions like what I've had with Foodhall. The other is the NHS [National Health Service] responders. They were lovely. They phoned and spoke to me and things like that. Cheered me up. (Diane, age 40-50)

For many other participants, interactions with volunteers were one of the highlights of their day; for those living alone, this was one of their main sources of social interaction. Some participants even expressed how these interviews were one of the longest conversations they had had in a long time. This shows how socially isolated people were during COVID, but also how important it was for people to maintain social interactions to ‘lessen the mental load.’

Community groups like Foodhall thus provided a critical service to people not just in the form of food, but also through the conversations had with volunteers and the wellbeing support offered. Randy, for example, used Foodhall’s wellbeing helpline service to speak on the phone to trained counsellors, which helped him cope with his depression. We thus see how community organisations like Foodhall can provide holistic support, but also create informal spaces for organic social interaction to occur. Randy, who first felt extremely ashamed to be using food support, later spoke about how the interactions he had even resulted in him making new friends.

> Made quite a few friends up at the Foodhall and up at the food bank...We've got news out of talking to new people through the Foodhall and through the food banks and that. Now everybody knows who we are up at the food banks and at the Foodhall. They know us all by the proper name and that. Every time I go up they're always asking, 'Are you okay? Is your family okay? Can we help with owt else?' (Randy, age 50-60)

As community groups and organisations create the space for people to build social networks, this also expands the resources available to people to cope with crisis. We know that social networks were an important source of information for people, as most participants found out about Foodhall through word of mouth. This is relayed as Randy explains how he "got news out of talking to new people" at Foodhall and the food banks he went to. However, the simple act of talking to others also eased the mental burden by being able to share about struggle. Having spaces where people can interact with others less close to them, or with mental health professionals, is extremely important for those who might not want to share with friends or family. We see the important role of volunteers in creating destigmatising situations where people felt comfortable asking for help.

5.2.4.2 Mutual aid and reciprocity

As participants began to realise that they were not alone in their struggles – indeed, many other people were facing similar challenges during this time – the neoliberal narrative and stigma that 'seeking help is wrong' began to change. As the 'norm' shifted, the idea of mutual aid began to become more commonplace, especially as this term was coined for the neighbourhood-based volunteer groups that were emerging across the country. As "mutual aid" entered everyday dialogue, participants began to internalise this concept: acknowledging that sometimes, they would be able to offer more support for others, and at other times, they may need more support. This idea of mutual aid was thus underscored by the ability to engage in reciprocal relationships. Reciprocity circumvented the stigma associated with 'giving someone charity' or a 'handout,' as participants participated in an exchange and found ways to 'give back.' This made asking for and accepting support for others easier as people began to feel a sense of community. Caroline expressed this as she described how people were actually "more helpful" and "more social" during COVID, which she saw as a positive.

> People seem to be more helpful because they seem to have realized that there are people who couldn't do things for themselves, and they've been a lot more helpful. The area where I live, people [before COVID] would walk past and just go, 'Hiya,' if they'd see me in the garden, but now they stop for a bit of a chat. 'Are you all right? Anything you want?' You know what I mean? They seem to be a bit more social, I suppose. I think everybody is feeling it. (Caroline, age 70-80)

Caroline attributes people's new behaviour to the fact that "everybody is feeling it", referring to the isolation, and also to the recognition that others needed help. Caroline thus identified that people do want to help others and that the feeling of 'being in it together' has helped people support each other.

This sentiment of solidarity underscores the idea of mutual aid and is further captured by Linda. She acknowledged the fact that sometimes she was in a position to help people, while at other times, like during the pandemic, she knew she needed to lean on others to get by.

> There's been times - many, many times in my life, I've been with nothing but the clothes on my back and my pets. And people dress you, people feed you. And you just have to count your blessings… And I'm very generous, you know, I'm a generous person in return. So, I just have to count on the kindness of strangers. (Linda, age 40-50)

Like Caroline, Linda acknowledges the good in people, describing how people that she did not know often came to her aid in the past. Linda further highlights the idea of mutual aid and reciprocity in how she recognises that sometimes she needs help from others now, but also is a “generous person in return.” Unlike many other participants, Linda did not experience self-stigmatisation and felt comfortable asking others for help. We saw that Linda had a strong sense of self-worth, confidence, and resiliency to overcome obstacles in her life, describing herself as “bright”, “highly educated”, and as someone who “manages the fully impossible.” Her recognition of mutual aid reduced barriers to accessing help from her personal support networks, circumventing the individualistic neoliberal stigma. Her optimism and sense of agency also facilitated her ability to engage in reciprocal relationships. This was seen in how Linda cooked meals for her neighbours in exchange for using their internet, and also in how she ‘gave back’ by cooking meals for NHS volunteers. As she engaged in these reciprocal exchanges of support, Linda was able to meet her needs while also helping others, contributing to her sense of agency and self-worth.

In contrast, most participants saw accepting help from friends and family as a last resort. However, there were some participants who, like Linda, consistently relied on help from their personal networks during this time. Examples included older participants like Caroline, Paul, and Lilly, who all relied on their (adult) children for support during lockdown. These participants did not express shame or embarrassment from having to rely on their children, in contrast to many of the younger participants this study, who often felt ashamed asking for help from friends and other family members. It can be hypothesized that the ease in which Caroline, Paul, and Lilly were able to accept help from their children was related to reciprocal exchange over a lifetime, as they had spent so much of their own lives caring for their children. On the other hand, younger participants mostly felt that they should be able to be independent and self-reliant, and thus did not want to have to rely on others.

In these cases where participants did not want to seek help from personal support networks, we saw how important it was to have other community options available. Although many participants did not even want to access these, because they did not want to feel as though they needed “charity” or a “handout”, they usually reported that they felt better once they finally asked for help. This is expressed clearly by Martin.

> I thought they [Foodhall volunteers] sounded brilliant because they just helped me out. They just said, ‘No problem. Any time you need help, just ring us and we'll help.’ I felt better for it as well, asking for help. I needed to ask for help. (Martin, age 50-60)

Although Martin had first expressed that he felt using food support was something he “shouldn't be doing”, he found that asking for help actually eased his burden and made him feel better. As friendly volunteers made him feel more comfortable in asking for and accepting help, we see that Martin came to shift his view of needing to ‘manage on his own’ to one where communal support and mutual aid was acceptable.

Foodhall understood the need to shift this narrative and aimed to do so by building community, providing a destigmatizing space and service, and creating opportunities for reciprocal exchange to occur through volunteer options. In these ways, Foodhall aimed to blur the traditional line between a ‘service provider’ and a ‘service user’ by encouraging those receiving support to take responsibility and ownership over the project. This builds personal agency and aims to erase the stigma of receiving charity or a ‘handout.’

Indeed, we saw that the desire for reciprocity was articulated throughout the stories of participants. All participants expressed their deep gratitude for Foodhall and the hopes that they could visit, volunteer, or donate to the project in the future. Some participants had engaged with community projects in the past, such as Gerald and Diane, who used to volunteer at food banks and community centres when they had been open prior to lockdown. William and Linda found ways to volunteer during lockdown by sewing masks or cooking meals for frontline workers. One participant even visited Foodhall to see about new volunteering opportunities after the interview, hoping to become more engaged with the community and put his woodworking and cooking skills to use. Finally, two participants – Martin and Gerald – both expressed that if they ever won the lottery, they would want to donate a large portion of their winnings to charities or community organisations like Foodhall.

> What I want to do is, if I ever become rich, whether it's from doing that with the betting or I find a decent job away from betting and that gets me rich, I always want to think about the people who've helped me along the way. Like the grassroots. Like yourself today. The other people who volunteer. I always want to remember them. I don't want to become big-headed, if I ever make it one day in the future. Some people forget about who's helped them. I think it's people like them [the volunteers] who's helped me to get where I am…I want to remember them and thank them all. (Gerald, age 30-40)

This shows the value that people held for these community organisations, and also the desire people had to give back, even if they were unable to at the time. This demonstrates a need to create avenues for people to support each other during times of crisis. Community groups and organisations like Foodhall can thus serve as an ultimate connector – providing a space to create communal networks and facilitate exchange in ways that re-build agency and control, during times when so much control is lost. In this way, collective spaces of care emerge that can rapidly respond to crisis by building on the strengths of the community and the idea of mutual aid.

5.2.5 Community food infrastructure for crisis response

We thus see how community organisations and spaces are vital for building resiliency in times of crisis through the ways that they connect people to provide mutual support. These community efforts were able to adapt quickly to ‘catch’ people that were left behind by other support options. These efforts were thus responsive to local needs, providing personalised and place-based services. Community groups like Foodhall understood the complex nature of food insecurity within their specific geography and worked to provide additional support to address the indirect barriers people were facing to accessing food, such as lack of internet access, mental health challenges, and stigma, when these were not necessarily recognised in national financial support and food delivery measures.

Community groups were also able to connect and collaborate with other local businesses to redistribute essential goods rapidly, reducing local waste and providing to those in need. Organisations like Foodhall thus become a crucial intermediary within a city’s food infrastructure, responding to surplus and local demand when national supply chains struggled to do so. These groups were thus essential in connecting people locally in a time when people felt more disconnected than ever. However, the informal and volunteer-based nature of these groups, which allowed them to rapidly adapt and respond to crisis, was set against long-term sustainability and consistency of support. As many of these organisations relied on grants, donations, and volunteer time, which began to dwindle after the first COVID-19 lockdown, the ability of these groups to provide these services in the long-term also waned.

5.2.5.1 Providing logistical support

One of the advantages of community-based support efforts were their flexibility and thus ability to quickly adapt to the local conditions of the COVID crisis. As an example, Foodhall was able to shift from being a community café that was open just three days a week prior to COVID into a free emergency food delivery service during the very first week that UK lockdown was announced; in just a few months, this service became one of Sheffield's largest locally-based food distribution efforts. This greatly contrasts the national government's inflexibility and slow response time, seen in how they were unable to adapt Universal Credit payments to come sooner for people or stop advance payment debts. Thus, we see that community efforts were essential to provide immediate support to people when they were unable to access other national services.

Having this option for immediate relief meant that Foodhall could 'catch' people while they waited to access other more formal services (e.g., benefits programmes, food bank referrals). As participants now had an option available which eased the worry of finding their next meal, this meant they could more readily focus attention on finding other consistent means of support.

For example, Linda expressed how Foodhall was a critical service that she relied upon when she could not afford food, which then allowed her to focus attention on pushing her visa application through government systems so she could apply for work and benefits again.

> [When I found out about Foodhall], I was sitting there one day after I had just been banging my head, trying everything I could do, not being allowed to work, can't get benefits. [sighs] I knew the visa [process] was gonna be long. (Linda, age 40-50)

We see that Linda felt helpless in her situation because of how difficult it was to access support for her visa, which would have provided her with the opportunity to apply for other state services. Foodhall was thus able to 'bridge the gap' for Linda, providing a 'meantime' service while she was applying for other help. In this way, Foodhall's service was able to take the stress off of the logistical struggles people were facing so they could focus on other challenges in their lives.

This was also seen by Lilly, an older woman who was living alone during lockdown. She expressed how, before she knew about Foodhall, she often found herself in situations where she had no food in the house.

> [Before using Foodhall's service] my cupboard was bare, but now I've got quite a few tins in, like beans and tomatoes and tins of soup. If my fridge is empty, I know I can have a tin of soup or something. (Lilly, age 50-60)

Foodhall's service provided Lilly with a 'safety net' of food in her cupboard, which meant she did not have to worry about going hungry if her own food ran out. Easing this stress improved her ability to cope with food insecurity by lessening the mental load. This is similarly expressed by Liam, who had been going through an extremely traumatic time at the start of lockdown. After splitting with his partner, leaving the family home, living homeless for a time, and then finally obtaining emergency accommodation, all whilst trying to manage his alcohol addiction, Liam explains how difficult it had been to cope with all these stresses. He further acknowledges how this stress is compounded by financial worries and his food insecurity.

> I never thought in a million years that I'd be able to cope living on my own. Although it's been an absolute nightmare and so emotional, at times, I think if it weren't due to money worries and food worries, being able to go out and get bits of stuff [so] that I can make stuff with what [food] I get given from yourselves [Foodhall], then I think I'd be all right. (Liam, age 40-50)

Liam explains that, despite all the challenges he has lived through during and after the first COVID lockdown, he has been able to work on his own mental health and cope with living on his own for the first time in over twenty years. He affirms that he feels like he would be "all right" now, if he just did not have to worry about money and where his next meal would come from. This again shows how critical it is to have support efforts that can catch people in times of crisis. Being able to ease the logistical struggles and stresses in peoples' lives before these challenges snowball to create further mental and physical health risks is extremely important, and thus requires fast-acting and adaptable services in times of crisis.

5.2.5.2 Addressing local and place-based needs

The further importance of having local, community crisis response efforts is that they can cater to specific place-based needs and challenges. For example, because of Foodhall's connection to the community, they understood the local needs that their emergency food distribution effort required. They were also able to work directly with other players in the local food system, which allowed them to provide a home for surplus food sourced from restaurants, hospitality and entertainment venues, shops, and farms and then quickly redistribute this food to those in need. These short supply chains reduced local food waste at a time when national and international food supply chains were unable to adapt to the changing supply and demand in the food system that resulted from lockdown measures (e.g., restaurants closing), which led to a large amount of food waste internationally (Yaffe-Bellany and Corkery, 2020; Filimonau, 2021; Roe, Bender and Qi, 2021).

Foodhall thus served as an intermediary in Sheffield's local food supply chain by receiving and redistributing surplus food, becoming a critical player in the city's food infrastructure. Their delivery service spanned across all postcodes in Sheffield (123 km^2), which was important as many areas, especially villages on the outskirts, lacked available services. Realising that internet access was an issue for many, Foodhall utilised a phone helpline system for food parcel requests, as phone access was more widespread at the time. Due to the small size and community nature of the operation, Foodhall could also provide a personalised service, calling each parcel recipient individually to see which items they needed and could actually use. A take-away parcel service was also put in place for those who could not receive deliveries to a permanent address. In these ways, Foodhall built their emergency food distribution effort to respond to local needs across the food system.

All participants expressed extreme gratitude for Foodhall's service, with many highlighting how important it was to be able to speak to volunteers and request specific items. A couple participants were receiving both government food parcels as well as Foodhall's parcels, which they found was necessary because the government parcels either did not provide enough food for the whole household or did not provide food that could be used because of dietary and cooking facility restrictions. Some participants required certain foods because of health conditions, for example, several study participants, including Kathy, were diabetic and thus needed to request specific foods.

> I’ve always said when they've [the Foodhall volunteers] come, ‘I appreciate you bringing us food.’ I said, ‘If it weren't for you lot, I think we'd be dead now.’ It's true, isn't it? If it weren't - I'm diabetic…I've got to have something for my sugar levels. (Kathy, 50-60)

We see how important it was for Kathy to know that there was a service that she could rely on to get the specific types of foods she needed. Although participants could not choose every food item in the parcels, Foodhall did make notes to not include certain items when parcel recipients mentioned that they did not want them. This helped reduce food waste in the household and ensure that people received foods that they actually could and would use. Paul, for example, details how helpful this personalised service was.

> I've got no qualms with yours [Foodhall’s service] at all. It was just a couple of things that you put in [that I didn’t like]. It said, ‘Mention if you don't,’ and I mentioned it, and I think they must have put it on a computer because they keep telling me, ‘Oh, you don't like so and so Mr. Pat.’ I said, ‘It’s not Pat, its Paul.’ ‘Oh right, sorry Paul.’ That's fine. I've never had them since, which has been brilliant. They've actually bent over backwards to help me. (Paul, age 60-70)

Foodhall was able to operate a personalised service because of the project’s local basis and smaller scale, which would have been much more complex to do in national food distribution efforts.

Other participants also expressed gratitude at the contents of Foodhall’s food parcels, often acknowledging that they appreciated both the mixture of foods and the inclusion of fresh foods, which was not usually given in government food parcels. Emily highlights this:

> The food bank in my area - I had to go and actually get the food parcel. I found that the Foodhall was actually giving me better quality food. I was getting fresh veg and fresh fruit and stuff. It was better quality food. (Emily, age 30-40)

Foodhall’s parcels were able to incorporate fresh produce because of their connections within the local food system. Foodhall received fresh produce from local shops and farms as well as freshly baked bread from local bakeries; because of the localised and short supply chains that emerged, Foodhall was able to deliver these fresh foods on either the same or next day.

The fact that Foodhall’s service required no proof of need and had both delivery and take-away options meant that their emergency food support was widely accessible to the city’s population. In particular, many participants spoke about how the delivery option made food support a more accessible option to them, as it lessened the financial stress of having to pay for transport to obtain food and overcame the physical barriers to accessing food for those who were self-isolating or who had trouble carrying food home themselves. William, an elderly man who lives alone, explained how Foodhall’s delivery service reduced his barriers to accessing food. William suffered from intense back pain from a prior work-related injury, which on bad days, prevented him from driving or from moving at all. Knowing that he could receive food deliveries thus lessened the stress he had around running out of food on a ‘bad day.’

> It’s brilliant, absolutely brilliant, because it [getting a delivery] means I don’t have to go out, I don’t have to go anywhere, I can just self-isolate. You know, I can drive, I have got a car, but if I go out it’s about once a month, or once a fortnight, that’s it - you know, for a few essentials... Foodhall has been a godsend really, love. It’s been a godsend, Foodhall. Because I’m not carrying as much weight back from the shopping. (William, age 60-70)

Other participants further confirmed how Foodhall’s delivery service removed stigma and made it easier to accept help, describing the service as a “godsend” because the food could be delivered to people directly, which was less “humiliating” than a food bank and “just like getting a delivery from Asda.” Thus, we see how important it was for community food groups to be able to address local and place-based needs; as these services recognised people’s specific challenges, they were able to build their services in ways that were more accessible and also lessened stigma.

5.2.5.3 Longevity and consistency

The informal and volunteer-based nature of Foodhall and other mutual aid projects at the time meant that they were able to adapt quickly to provide almost-immediate support to people; at the same time, this grassroots structure also meant that these projects faced issues with consistency and sustainability in the long-term.

Indeed, Foodhall’s emergency food distribution operation ran in full capacity from March through August 2020. After this, they operated a much smaller delivery project, and focused efforts on re-opening their café and signposting people to other options for food support. Re-opening the café allowed Foodhall to realign with their mission of bringing people together around food. Indeed, operating an emergency food distribution service had never been Foodhall’s mission or role in the community; however, due to their non-hierarchal structure, they were able to adapt to meet this immediate local need during the first lockdown. Thus, we see that even though Foodhall never planned to maintain a large-scale food parcel distribution service, the fact that they were able to adapt and fill this role during a time when so many needed food support highlights the importance of having these types of community spaces as critical parts of a local food infrastructure. At the same time, if the role of these more informal and grassroots organisations are not recognised and made sustainable, it also means a loss of local resources that are able to shift in times of crisis to address place-based needs.

Indeed, Foodhall suffered a major financial hit from providing food parcels during the first COVID-19 lockdown. Despite the fact that Foodhall received a wide range of food donations from local shops, restaurants, farms, and even from the city council during the first COVID lockdown, the organisation was still buying in a large amount of food themselves – approximately 70% of the total food supply. This expense coincided with a loss of traditional sources of income for the project, such as from events and workshops, leaving Foodhall in a state of financial crisis that still affected the organisation years later. It is thus clear that there is a need for consistency and reliability in these services, which can be difficult because of reliance on grant funding and voluntary support.

The role that these community spaces play in connecting people providing support has been evidenced throughout participants' stories. As many drop-in community centres, mental health services, and support groups closed during COVID, we saw how participants struggled to cope with their stress. Gerald's story showed how the closure of the community groups and centres, which he attended every day before COVID, diminished his options for daily activity, leaving him struggling to cope with his anxiety and gambling addiction. Gerald mentioned how the inconsistency of these groups, which would often operate in different churches and community centres throughout the city one time a week, was already 'not enough' even before COVID.

> They [the day centre] do that [meals] and there's table tennis, there's pool, there's snooker in the main hut. They do quizzes and other stuff. I asked him [the person in charge], "Can't you just put this on for seven days a week?" He said, "If it were up to me, I would, but it's up to the church." I know they've got other things on as well. One day a week just didn't seem enough. Two or three days a week would have been about right, ideally… Yes, the day centres were great, not just on a Tuesday. I've been to other ones, as well. I was going to one every day [before COVID]. Obviously, since the pandemic, I haven't gone to any. It's crazy. When are we going to get out of this mess?" (Gerald, age 30-40)

We see how Gerald relied on these day centres as a source of daily activity and social interaction, which he asserted was crucial for his mental health. Gerald's favourite group (the 'Tuesday Club') was only put on once a week, in a church. He was still able to go to other day centres across Sheffield on other days, but he would sometimes be walking over an hour to reach them, something that may not be a possibility for everyone. We thus see how important it is to have these types of community-based and social spaces as consistent, reliable services in people's own locality. COVID disrupted the viability of spaces like these, leaving many, like Gerald, on their own with no support. Although some community groups like Foodhall were able to shift to provide other services during this time, that action in itself threatened the longevity of the project.

It becomes clear that financial support for these community organisations spaces is essential. These organisations can adapt during crisis to provide immediate logistical support in line with local needs, easing the burden so people can tend to other life challenges. These spaces provide an opportunity for people to come together and support each other, thus building personal networks and community as resources to draw upon in times of need. However, as these spaces are placed into the 'third sector', instead of being supported as a critical part of a local infrastructure, the ability to respond to future crises may be threatened.

5.3 Discussion

In this study, the voices of fourteen participants have been foregrounded to capture their lived experiences of food insecurity during COVID-19. Key themes have been inductively derived from the data, as the aim is to draw conclusions grounded in the words and experiences of participants as they have reported them. From these interviews, three overarching and interrelated themes emerged, as participants described their situation as "stretched", "struggling", and "stressed." These three themes relate to the how the resources available to participants to achieve food security were stretched, resulting in newfound struggle to find ways to cope and exacerbating the mental load for participants through emotional and physical stress. We see how each of these themes diminished the resources available to participants to achieve food security and created new barriers to then coping with food insecurity. Also drawing upon participants' words, we find that what helped participants most was people – both within personal networks and also within community organisations, thus showing the importance of social capital as a crucial resource to achieve food security. Combining the qualitative data from the interviews with information about Foodhall's emergency food response during COVID, it becomes clear that community efforts were not only important but necessary to ensure that people could meet their basic needs during the pandemic.

This research thus highlights the resources that were critical to achieve food security in the face of crisis. Drawing upon the words of participants, a 'resource map' has been derived, as portrayed in Figure 73, showing the resources utilised to achieve food security and cope with food *in*security. This framework provides evidence that the resources needed to achieve food security are much more than purely financial. Other individual and family-based resources, such as mental and physical capabilities, as well as the availability of social and place-based resources, are extremely important to maintain food security in the face of crisis, thus contributing to resiliency. These resources are interconnected in such a way that exerting strain on one resource or capability increases pressure on all others. The way that these resources intersect determines the options of food and support available to an individual in a specific local context.

Utilising this resource map as a framework, I then provide the following major contributions within this discussion. First, I highlight the multi-dimensional nature of household food insecurity, showing how this is intricately linked to mental and physical health in a compounding and reinforcing cycle. I demonstrate how the resources necessary to achieve food security were diminished as a result of COVID-related struggles and lockdown measures, but provide further context to how these resources were shrinking long before, linked to neoliberalism and austerity policies. Within this realm of shrinking government support, people showed immense capability and resiliency as they aimed to stretch their resources, but often this was just not enough. This resulted in immense stress which furthered mental health challenges, again compounding upon the experience of food insecurity. The ways that neoliberal-ableism was internalised by participants created further barriers to receiving the support they needed, as this embodied stigma resulted in participants relying on themselves to make things work until reaching out for external support became the last and only option available.

I further show how the experience of food insecurity is linked to other resources both inside and outside the household, such as family, social networks, and place-based (e.g., community) resources. I explore how community-based responses to the COVID crisis, built upon ideals of reciprocity and mutual aid, were vital to overcoming the stigma-based barriers in accessing support and also in providing fast-acting support for people at the time they needed it. These responses were thus crucial for both individual and community resilience in the face of crisis. Drawing this together, I argue for a community food infrastructure that includes community food spaces like Foodhall, connected to other local food actors and health services, to develop the critical social and place-based resources needed to build resiliency to future crises.

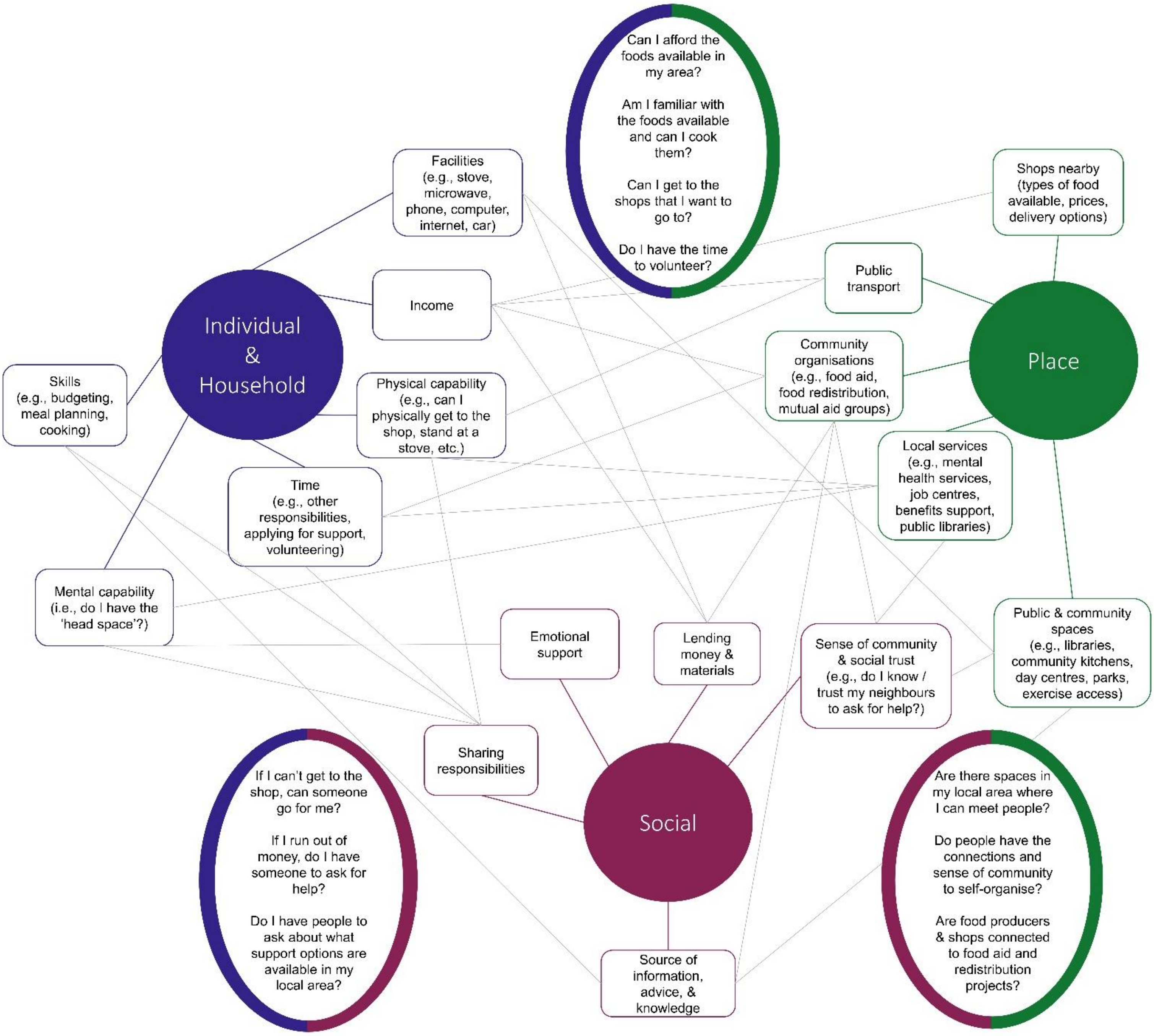

Figure 73 *- Food security resource map. This framework shows the resources drawn upon to achieve food security and cope with food insecurity by participants within this study (Chapter 5) during the COVID-19 pandemic. These resources are characterised as individual / household, social, and place-based. Resources are interconnected, both within and between groups, which is demonstrated by the linking lines drawn between them. Thus, when strain is exerted upon one resource, increased load will also be placed on the others. The ways that these resources intersect determines the food and support options available to an individual within a local context, with examples provided within the ovals between groups.*

5.3.1 Food security: A system of household, social, and place-based resources

Drawing upon the words of participants, we find that the resources needed to achieve food security extend beyond an availability of local food and the means to access it. Mental and physical capabilities to utilise food, underscored by mental and physical health status, are also critical determinants of food security. Additionally, the availability of resources that one can draw upon in times of crisis, even if these resources are not needed all the time, are necessary to ensure that food security can be achieved *consistently* over time. This means that the resources needed to achieve food security must also include those that enable resiliency in times of crisis. COVID made this clear as the crisis impacted people across socioeconomic lines, and even those financially able often had to rely on friends, family, and community organisations for help in accessing basic essentials.

Figure 73 provides an overview of the individual, household, social, and place-based resources that participants drew upon to address the multitude of challenges that surrounded their experiences of food insecurity during COVID-19. We see that these resources are intricately connected to each other; both within their respective groups and in between them. These resources thus form a complicated and interconnected system, which provides context for why the experience of food insecurity is much more than just an issue of affordability or physical access.

I frame the connection of these resources presented in Figure 73 as a metaphor. Consider a heavy plane of wood that is balancing a glass of water, held up by many skinny beams – essentially a table with many legs. In most cases, a table only needs a few legs. In viewing these ‘legs’ as someone’s individual resources and capabilities, we see that in times of stability, only some resources are needed; for example, having enough money negates the need to spend time travelling farther to budget supermarkets. However, other resources are needed when crisis strikes and key resources are diminished. As one ‘leg’ breaks, the load will be transferred to the others; having more legs under the table lessens the pressure on each leg and prevents the glass from falling. In the same way, we see that as one’s personal resources are constrained, this exerts more strain upon other resources. For example, as income was diminished for participants, they drew more upon budgeting and meal planning skills and took more time to work out how to meet their various needs – something that many did not do before COVID. The availability of a wide array of resources means that the strain exerted upon others may not be so severe when one is lost.

Continuing this metaphor, we see that drawing upon social and place-based resources expands the resources available to an individual beyond their own skills, time, capacities, and income. This creates more legs under the table, or in some cases even allows for the glass to be transferred to another table entirely as the load of a logistical burden is shifted. This was seen, for example, when community efforts helped get food and other basic necessities to people during COVID, negating the need for them to find out how to make this work by themselves. However, just as resources can be expanded by others, they can also be narrowed in contexts that attack many ‘legs under the table’ all at once, like COVID; in some cases, the depletion of these resources can cause the system to collapse entirely and the glass to shatter, leaving people in a severe state of food insecurity. Trying to simultaneously manage many challenges all at once (rebuild many different legs) with a stretched and diminished set of resources can make it very difficult to 'put the glass back on top', thus limiting the ability to find ways to cope with food insecurity (Sen, 1990).

5.3.1.1 COVID-19: An attack on all sides

Framed within this context, we see that the COVID-19 crisis was an 'attack on all sides', depleting a multitude of the resources needed to achieve food security all at once. COVID created barriers to all four dimensions of food security as defined by the FAO (2008): food supply, access and affordability, utilisation, and stability of these over time (Lambie-Mumford, Loopstra and Gordon, 2020). The lack of affordable food in local shops, shielding and self-isolation requirements, and newfound financial struggle created intense barriers to obtaining food, whilst also negatively affecting physical and mental health and thus further compounding the experience of food insecurity. This vicious and reinforcing cycle was played out within a context of minimal support for people, narrowed by over a decade of austerity and then further erased as many public spaces and community centres that people had depended on were now closed during lockdown.

The direct barriers to accessing food were mainly financial and physical; participants struggled to afford food and could not physically access it, often due to the need to shield or self-isolate. Financial struggle was related to issues such as loss of work; inadequacy of benefits; increased expenses, related to utilities (more people in the house) or food (having to rely on more expensive local shops); and changing household income (e.g., as people moved out for lockdown). These struggles were compounded by the insufficient supply of affordable food in one's area and the inability to obtain supermarket delivery slots, which often meant that people were now spending more money on food at smaller, local shops or relying on someone else to do so. These physical and financial barriers are consistent with other surveys and studies on drivers of UK food insecurity during COVID-19 (Lambie-Mumford, Loopstra and Gordon, 2020; Loopstra, 2020; Bramley *et al.*, 2021; Sheffield City Council, 2021; The Food Foundation, 2021).

However, what this research uncovers is that the challenges people faced and their experiences of food insecurity were much more complex than just a lack of food on supermarket shelves or a lack of money to afford it. As also evidenced within prior research exploring the lived experiences of food insecurity and poverty (Shildrick, MacDonald and Furlong, 2016; Rose and McAuley, 2019; Sosenko *et al.*, 2019), for participants in this study the experience of food insecurity was unique, complicated, and multi-dimensional. Food insecurity was weaved within other issues associated with changing households, social isolation, mental and physical illness, severed relationships, and traumatic life experiences. The multitude of hardships that participants experienced at this time built on and fed into each other, compounding, accumulating, and complicating daily life. Shildrick, MacDonald and Furlong (2016) uncovered this multiplicity of hardship for families living in poverty, describing how troubles "come not as single spies, but in battalions."

COVID in particular spurred this multiplicity and compounding nature of hardship, because it changed not just one aspect of life but redefined 'normal life' in and of itself (Burn and Mudholkar, 2020). In this study, this was seen particularly in Gerald's story, as the structure of his day was completely redefined when the community centres that he normally went to for a hot meal, social interaction, and physical activity closed during COVID. The loss of this led to sleeping troubles, heightened anxiety, and difficulties in managing his gambling addiction, which furthered his financial struggle and ability to afford food. Difficulties in managing addictions and mental illnesses were seen within other participants' stories, as well as other studies during this time, often related to the stress surrounding COVID, household tensions, and changes in daily structure (Burn and Mudholkar, 2020; Marsden *et al.*, 2020). We thus see that affording food was only one challenge faced by people during COVID; the need to find ways to cope with the many other life challenges spurred by the pandemic, at a time when people were physically isolated from their friends and family, presented an intense mental load that made it harder to cope.

5.3.1.2 A narrowing of government support

Many of the resources available to a particular individual to achieve food security – or simply to "manage" – were wiped out by the global pandemic. However, this research also shows that many of these resources were already being constrained or diminished prior to COVID. Indeed, food insecurity did not just become an issue during the pandemic - it just became more widespread (Sosenko, Bramley and Bhattacharjee, 2022).

In particular, the stories of participants show how the pervasion of neoliberalism into the UK political and social sphere has narrowed the resources available to people to cope during crisis; fitting into the prior analogy, this can be seen as a removal or a weakening of the legs under the table. The dominance of austerity in UK politics in the decade or so leading up to the pandemic has created a dismal and barren landscape of support (Wood, 2012; Rose and McAuley, 2019; Jenkins *et al.*, 2021), which meant many did not have any kind of 'safety net' to fall back on during the shock of crisis. Indeed, many participants who were reliant on benefits before the pandemic were already struggling financially when COVID hit. The inadequacy of these benefits was expressed as one participant explained "you just don't get enough to manage on." The Trussell Trust foodbank network has consistently cited that the most significant driver of UK food bank use, both before and during COVID, is the insufficient income provided by the social security system (Bramley *et al.*, 2021).

It is harrowing to see the similarities between the stories presented within this research and the stories of families previously affected by austerity measures, starting with the nearly £18 billion worth of cuts to the UK benefits system that came with the 2010 Coalition Government (Wood, 2012). The reductions in benefits income and available services that came with austerity policies have been linked to rises in food insecurity and food bank use throughout the last decade (Douglas *et al.*, 2015; Jenkins *et al.*, 2021). This has also led to declining mental health for those impacted, characterised by increased anxiety and depression, constant physical and emotional stress, social isolation, an inability to participate in everyday life, and excessive shame and stigma (Wood, 2012; Pemberton *et al.*, 2017; Mattheys, Warren and Bambra, 2018; Rose and McAuley, 2019). This was all before the context of a global pandemic.

For those who had been 'just getting by' prior to COVID, the pandemic was then like a metaphorical flood that wiped the legs out from under them. In this study, we saw that Diane could only make it through three weeks of the month before her Universal Credit payments ran out, all whilst loans were still being deducted from her benefits during lockdown. The continued subtraction of these advance debt payments, as well as the minimum five-week wait time required to receive the first Universal Credit payment, were consistently cited as major drivers of financial struggle and food insecurity during COVID (Economic Affairs

Committee, 2020; Power *et al.*, 2020; The Trussell Trust, 2020b; Trades Union Congress, 2020; Work and Pensions Committee, 2020a, 2020b; Bramley *et al.*, 2021; Patrick and Lee, 2021).

This inadequacy of benefits sat atop an unavailability of mental health services for people, due largely to how stretched health services were at this time and the difficulty of accessing changing formats of care provision (Iob, Steptoe and Fancourt, 2020; Giebel *et al.*, 2021; Molodynski *et al.*, 2021). Several participants in this study struggled with mental illness, and several more reported their mental health worsening during COVID, but most were unable to access any mental health support during this time. Previous studies have recognised that the availability and affordability of mental health and disability support services were already decreasing before the COVID-19 crisis, due largely to austerity policies (Wood, 2012; Reeves *et al.*, 2013; Goodley, Lawthom and Runswick-Cole, 2014). This meant that when crisis struck, it was difficult for these already stretched services to cope with new demands for support.

The Government did commendably increase expenditure on a massive scale to support people during COVID, through measures and programmes such as the £20 per week uplift in Universal Credit, income support for those furloughed or self-employed, vouchers for families with children normally in receipt of free school meals, and food parcel deliveries for those shielding (Lambie-Mumford, Loopstra and Gordon, 2020). However, we see that these reactionary measures were still not enough for many, and often people were left hanging in the 'meantime' while they waited for support to come through. Some 'fell through the cracks' of these support programmes entirely and were left dangling with few options for help outside of local community efforts (Robertshaw *et al.*, 2022).

This was also seen for participants in this study. For example, Linda could not receive any benefit support following unemployment due to the extreme delay she experienced in renewing her visa. Paul was left shielding without food when the government food parcels that he was entitled to stopped coming to his house with no explanation. Other studies have also highlighted how difficulties in navigating application systems for support, system delays, and confusion about eligibility for various programmes created significant barriers in applying for and receiving support during COVID (Blundell and Machin, 2020; Bramley *et al.*, 2021; Robertshaw *et al.*, 2022; Sosenko, Bramley and Bhattacharjee, 2022). Time and time again people are being left behind as cuts are made and 'there is no one there' to help. Although these issues worsened with COVID, they were already present before; too many stories of struggle during times of austerity were the result of bureaucratic errors, as benefits for many stopped with no explanation or came with significant delay, with many finding difficulty in 'talking to the right person' to resolve these errors (Wood, 2012; O'Hara, 2015; Pemberton *et al.*, 2017). Thus, we see that although the Government did allocate massive funding to support people during COVID, the previous decade of austerity had already ground away at the resources available to people to cope within a crisis. COVID only made this harder as many of the local services and spaces that were available to people before now had to close their doors with lockdown.

5.3.1.3 Utilising individual resources to cope

The inadequacy and lack of a 'safety net' of support for people during this time left participants relying on their own capacities to get by. Participants first drew upon individual or household resources to address newfound struggle, regarding external support from personal networks or food aid as a last resort, as commonly seen in past research (Hamelin, Beaudry and Habicht, 2002; De Marco, Thorburn and Kue, 2009; Purdam, Garratt and Esmail, 2016; Middleton *et al.*, 2018). As individual resources were stretched and participants were placed under extreme stress to find new strategies to make things work, both physical and mental health were impacted in such a way that the resources and capabilities available to participants were further diminished. An embodied neoliberal-ableism further narrowed the resources available to participants to cope with food insecurity and also exacerbated this experience through the stress and health challenges that it created.

The strategies used by participants to cope with new financial and physical barriers to accessing food required significant time, effort, and skill. Participants found inventive ways to reduce costs, created meticulous budgets, and drew upon organisational and meal planning skills to stretch their resources, as commonly seen in prior research examining coping strategies to food insecurity (Hamelin, Mercier and Bedard, 2010; Douglas *et al.*, 2015). In this study, we saw this in Linda's careful meal planning, her intense organisation in re-submitting her visa applications every five weeks after they expired, and the eight hours a day she spent calling multiple offices to find support options. This strength and tenacity often seen within times of extreme struggle is what Zipfel *et al.* (2015) refer to as the 'powers of endurance.'

However, despite the wide-ranging 'powers' displayed by participants, these strategies were still not enough cope. Participants were thus forced to make heart-breaking calculations between their various needs. As evidenced in prior research, food was commonly a place where cuts were made (Dowler, Turner and Dobson, 2001; Dowler and Lambie-Mumford, 2015; Pemberton *et al.*, 2017; Blake, 2021; The Food Foundation, 2021), often so that participants could instead afford rent, utilities, phone service, or in some cases, transport to see their children. Participants stretched foods out into several meals, 'traded down' on the quality of meals, or skipped meals entirely to afford other needs or to ensure other family members could eat; these strategies – or better termed sacrifices – are consistently reported among those experiencing food insecurity (Cooper, Purcell and Jackson, 2014; Dowler and Lambie-Mumford, 2015).

5.3.1.3.1 Links between food insecurity and health

There is an immense amount of physical and mental effort that goes into coping with crisis and stretching minimal resources (Holman, 1998; Douglas *et al.*, 2015; Dowler and Lambie-Mumford, 2015; Purdam, Garratt and Esmail, 2016). However, employing these strategies can also take an immense physical and mental toll. One case of this is seen in how the diets of participants changed drastically during COVID, especially for those most reliant on food support. Participants reported eating mainly meals like "beans on toast", "a tin of soup", "a jacket [potato] with nothing on," or even "a tin of tomatoes, mushy peas, [and] garden peas." Past research has shown how the stress and poor diets associated with food insecurity can lead to physical illness (Dowler, Caraher and Lincoln, 2007; Berlant, 2011; Gundersen and Ziliak, 2015), which can therefore affect the capability to be food secure in the future. Although Foodhall was usually able to provide participants with fresh produce, eggs were an occasional 'treat' and meat was not usually an option. In many cases, food aid organisations may not be able to provide fresh items, and when they do, there can be concerns about food quality (van der Horst, Pascucci and Bol, 2014; Middleton *et al.*, 2018). We therefore see that the decisions that people have to make to survive day-to-day in times of hardship – and the stress associated with making them – may come with a great cost for their health and thus their future (Dowler and Lambie-Mumford, 2015).

The crushing stress and frustration in trying to manage on limited and inadequate resources was expressed by many participants in this study. The need to piece together a network of temporary solutions – previously described as a 'patchwork of provision' (Rose and McAuley, 2019) – requires extreme effort and leads to mental exhaustion (Holman, 1998; Zipfel *et al.*, 2015). The unpredictability and insecurity that people experience as they move from one challenge to the next creates a state of constant worry (Pemberton *et al.*, 2017). This is set within a context of diminished state support and the loss of the 'safety net' through austerity policies (Reeves *et al.*, 2013), which means that when crisis hits, people find themselves in dire situations before help is found (Wood, 2012). In this study, this was seen in how Randy had to skip meals before he found out about Foodhall, and Linda was already months behind on rent before she got help with food.

It therefore becomes quite clear how the constant struggle of food insecurity and daily stress of finding ways to cope negatively impacts both mental and physical health (Gundersen and Ziliak, 2015; Rose and McAuley, 2019; Pourmotabbed *et al.*, 2020). The interaction between mental health and food insecurity – or even poverty more generally – has been previously evidenced as a bi-directional relationship (Lund *et al.*, 2011; Tarasuk *et al.*, 2013; Martin *et al.*, 2016; Noonan, Corman and Reichman, 2016; Ridley *et al.*, 2020). In this way, mental illness can also create barriers to achieving food security, as mental capability is hindered and the resources available to cope with crisis are further narrowed (Heflin, Corcoran and Siefert, 2007). In this study, this was seen in how William's anxiety, spurred from a violent attack that happened to him thirty years ago, prevented him from going to the food bank because of the number of people there and the stress that he associated with the experience. Other mental illnesses can also pose a barrier to organising and planning ahead in life; depression, for example, is often characterised by difficulties in getting out of bed and feelings of being stuck or helpless (Holman, 1998; Rose and McAuley, 2019). Coping with illnesses like depression and anxiety is mentally taxing (Ridley *et al.*, 2020); this pressure, atop other financial and health challenges, leads to immense stress and mental exhaustion, which makes it harder to implement the various time- and effort-intensive coping strategies usually executed during times of food insecurity (Heflin, Corcoran and Siefert, 2007; Tarasuk *et al.*, 2013). This mental exhaustion in trying to stretch resources is poignantly captured in an interview by Pemberton *et al.* (2017), where one individual experiencing financial struggle stated: "People say money burns a hole in your pocket – I say it burns a hole in my head."

Within this study, the mental toll of food insecurity and the further barriers to coping that this presented were particularly seen in Kathy's case. Suffering from bipolar disorder and a recent brain haemorrhage, Kathy had lost a large portion of her household income when her son, previously receiving a carer's allowance for her, moved out. Complications within the state benefit system therefore left Kathy in a dire financial state; unfortunately, the ways in which these errors or inflexibilities in the benefit systems drive food insecurity are not new (Dowler and Lambie-Mumford, 2015). Kathy then faced further challenges in remedying the situation. Unlike Linda, who spent eight hours a day calling support offices and government services to sort out her visa and benefits, Kathy was unable to do this because she often did not have enough phone credit and also struggled with her memory. We see that Linda had to draw upon multiple resources – time, knowledge, skills, and social connections – to find ways to cope and access support options. For Kathy, the difficulty in drawing upon these resources due to both physical and mental illness left her in a severe state of food insecurity – sometimes having to eat out of bin when she had no food left. Kathy's situation was set within a landscape of minimal state support, as she expressed difficulty and frustration in trying to access mental health services amid lockdown. Her story thus provides context to the many statistics on increased risk and rates of food insecurity for people with disabilities or ill-health, including mental health (Tarasuk *et al.*, 2013; Schwartz, Buliung and Wilson, 2019; The Food Foundation, 2021).

The exhaustion of life in struggle and intense mental load this brings can be related to the concept of 'slow death', as typified by Berlant (2007) in the context of waged workers in a neoliberal capitalist setting. 'Slow death' was characterised by a drawn-out exhaustion or "wearing out" of waged workers, who experience more physical and mental health illnesses due to their work and are thus dying more (albeit slowly), in contrast to workers on higher incomes (Berlant, 2007). Goodley, Lawthom and Runswick-Cole (2014) extend this 'slow death' to people with disabilities, arguing that the diminished resources and support available to people with disabilities because of austerity measures will lead to premature deaths. In this study and many others, the exhaustion, frustration, and stress that characterises the experience of food insecurity and financial struggle is clear; for most participants (12 out of 14) this was layered atop a physical or mental health challenge. The driving links between food insecurity, physical illness, and mental illness mean that these experiences compound upon each other, snowballing and tunnelling people into more and more severe states of ill-health and destitution – which can again be seen as a 'slow death.' However, what we find is that during times of crisis, this 'slow death' is not slow at all. It is the grim case that COVID brought this to light, as the risk of dying from the virus was higher for those who were food insecure or had diet-related illnesses commonly associated with food insecurity, as well as those on lower incomes (Bhatia, 2020; Yancy, 2020; Elliott *et al.*, 2021; O'Hara and Ivanic, 2022).

5.3.1.3.2 Stigma as a barrier to support

The stigma associated with being food insecure was also seen to have implications for mental health, thus straining the resources available to participants to cope with crisis. Participants' sense of autonomy was called into question when they found that their individual capabilities were just not enough stretch their resources, which for some eroded self-confidence and self-worth. As participants felt 'forced' to turn to food support, having to rely on others to meet a basic human need left many with even more feelings of shame, embarrassment, lost pride, stress, frustration and even helplessness, as has been evidenced in a range of other studies on food insecurity (Tarasuk and Beaton, 1999; Hamelin, Beaudry and Habicht, 2002; Hicks-Stratton, 2004; Loopstra and Tarasuk, 2012; van der Horst, Pascucci and Bol, 2014; Douglas *et al.*, 2015; Garthwaite, Collins and Bambra, 2015; Middleton *et al.*, 2018).The loss of identity as someone who could provide for themselves thus led many participants to stigmatise themselves in their own minds.

The deep desire and need to be able to 'manage on your own' was expressed by many participants and reflects an embodiment of the neoliberal ideal of personal responsibility

(Wrenn and Waller, 2017). As participants no longer fit within the neoliberal-ableist 'norm' of an able-bodied, autonomous, and self-sufficient individual, they came to view themselves as 'less than' or as an outsider in society. This individualistic neoliberal rhetoric has long pervaded everyday life, seen with the stereotyping of those who receive benefits or use food support as 'shirkers', 'scroungers', or just 'lazy' (Pemberton *et al.*, 2017; Rose and McAuley, 2019). This stigma is grounded within a neoliberalist narrative that exalts human choice and agency, supposing that a person's struggle or inability to overcome challenges are related to personal traits and behaviours (Sen, 1999b). Indeed, many past policies addressing food insecurity in high-income countries have been targeted based on this 'behavioural or capability deficit', viewing the food insecure as lacking in cooking, meal planning, or budgeting skills and capabilities (Dowler and O'Connor, 2012; Douglas *et al.*, 2015). This is of course contrasted by the strength, capability, and tenacity portrayed by participants in this study, who deployed a wide range of skills and resources to overcome their struggles. This begs for a viewing outside of the neoliberal narrative, by recognising the role and power held by societal structures in shaping the choices that are actually available to people (Sen, 1999b; House, 2008). Despite the ways in which these choices were diminished and erased by external circumstances outside of participants' control – namely, a history of austerity and the crisis of a global pandemic – participants still individualised their burdens, viewing their struggles as personal failures.

Oftentimes, what most profoundly affected the participants in this study was the fear they held of being stigmatised by others, or indeed the ways in which they stigmatised themselves, in contrast to actual instances of being stigmatised by others; this self-stigmatisation or fear of being stigmatised has been seen within a range of studies on food bank users, as evidenced in a recent review by Middleton *et al.* (2018). In this study, this self-stigmatisation was seen in how Caroline viewed her need to rely on food aid as a loss of independence and a personal failure, even when she did not hold this view for others using the same services. We saw that this embodied stigma often presented a barrier for participants in reaching out for help, with many waiting until the last possible moment before seeking external support – once their individual capacities had been stretched to a breaking point and they could no longer "survive" otherwise. Past studies have similarly shown how stigma – either actual, perceived, or embodied – has created barriers to asking for help from friends and family (McNeill, 2011) or accessing food aid (van der Horst, Pascucci and Bol, 2014; Caplan, 2016; Purdam, Garratt and Esmail, 2016; Power *et al.*, 2018), with these options again being viewed as a last resort. The state of desperation and lack of other options that leave people feeling 'forced' to reach out for food aid was expressed in this study and has also been well-documented among food bank users (Hicks-Stratton, 2004; De Marco, Thorburn and Kue, 2009; McNeill, 2011; Runnels, Kristjansson and Calhoun, 2011; Perry *et al.*, 2014; Douglas *et al.*, 2015; Middleton *et al.*, 2018).

Stigma not only prevents people from seeking formal support options ,such as applying for benefits, accessing mental health services, or going to food banks, but also isolates people from their own personal networks. As people view themselves as different to a societal norm, this leads to feelings of aloneness as they see themselves as sitting 'outside the box' of normal society. Past studies have also shown how shame and stigma can lead to detachment from society and possible sources of support (Lutwak and Ferrari, 1997; Chase and Walker, 2013). People often aim to hide their situation from others for fear of being stigmatised by those closest to them, which causes further isolation (Goffman, 1963; Pachankis, 2007). We saw this in the way that Liam hid his homelessness and financial struggle from his family because he did not want anyone to change their perception of him as a “as a strong person that has never needed anything in his life,” and the way that Gerald hid his anxiety “because people don't always believe you.” This isolation and the associated stress related to the fear of being ‘discovered’ can further increase the risk of mental illness, contributing again to the vicious and reinforcing cycle between mental health and food insecurity (Martin et al. 2016; Blake 2019). It thus becomes clear that individual capabilities and resources are intricately linked to each other and impact each other. As one resource became diminished – such as income or being able to access food by oneself – this exerted intense strain on all other resources, in such a way that physical and mental capabilities were further diminished under this increased load.

5.3.1.4 Family and household resources

The experience of food insecurity for participants was in many cases linked to others in the family or household. Past research has long shown how food insecurity can be experienced differently in families and also mediated by family members, such as in the way that parents skip meals so their children can eat (Tarasuk, McIntyre and Li, 2007; Power *et al.*, 2018; Sosenko *et al.*, 2019). Individual situations can also influence the whole household; this is seen in the way that the risk of food insecurity for a household is increased if there is just one member with a disability or chronic illness (Tarasuk *et al.*, 2013; Higashi *et al.*, 2017). On the other hand, in this study we also saw how living alone created barriers to being food secure, especially for those shielding, as they did not have someone readily available to shop for them. This research thus extends the idea of familial experiences of food insecurity to show how the presence or loss of household members expanded or diminished the resources available to achieve food security.

As household compositions changed just prior to COVID, some found themselves supporting an increased number of people, while others found themselves now living alone and needing to meet household expenses with a reduced income. Having family or others within the household could increase costs, but also expand resources, as food, skills, money, capabilities, and responsibilities could be shared (De Marco, Thorburn and Kue, 2009; Higashi *et al.*, 2017; Biroli *et al.*, 2021). In this study, we saw how children shopped for their elderly parents, as in Paul and Lilly’s case, and how parents helped to financially support their adult children during this time, as done by Caroline. In many cases, having people within the house could contribute to better mental health during COVID by having an option for social interaction and emotional support during lockdown (Bu, Mak and Fancourt, 2021; Kwong *et al.*, 2021). Stress could also be reduced by the ability to share responsibilities and tasks; indeed, Biroli *et al.* (2021) found that couples in the UK, U.S., and Italy who shared household tasks during lockdown reported the lowest levels of household tension.

Contrastingly, living alone could narrow the resources available to people. Financially, this was seen in how Kathy lost the carer’s allowance her son received for her when he moved out just before lockdown. Similarly, living alone could also mean a loss of capabilities and skills in the household. Now that Kathy was living alone and shielding, she had to pay someone to shop for her because she no longer had someone else in the house who could do so. Liam, living alone for the first time in over twenty years during the pandemic, struggled to budget, pay bills, and meal plan, tasks that had previously been completed by his partner.

In other cases, lockdown restrictions could incite tense situations within the household or within families. Liam's relocation was a direct result of his split with his partner, who had asked him to leave due to difficulties in managing his alcohol addiction. Liam attributed this split partly to the tense household situation spurred by everyone needing to be in the house all the time during lockdown. Other reports have also mentioned that the change in the normal structure of the day for people, atop being in the house with others all the time, led to difficulties in managing addictions and to a dependence on unhealthy coping mechanisms, such as drinking (Burn and Mudholkar, 2020). In extreme cases, we also saw how tension, stress, and increased time inside the home could result in conflict and violence, seen in the surge of domestic violence and child abuse cases during lockdown (Gibson, 2020; Ellis *et al.*, 2021; Molodynski *et al.*, 2021).

We thus see how others in the household could prove to be a crucial support system to cope with crisis, but could also diminish the capability to cope when people moved away or situations became tense and unmanageable. Although familial ties and the context of the household are important to food security – perhaps why 'household food insecurity' has been coined as a major research topic – the capability to achieve food security, especially in crisis, is also linked to the resources available *outside* the household. As participants' individual and household resources were stretched, but still not sufficient to cope day to day, they began drawing upon external sources of support that were available through their personal and community networks.

5.3.1.5 Social and place-based resources

We thus extend beyond the individualistic neoliberal narrative of personal resources and capabilities to show that social and place-based resources are also necessary to achieve food security, as depicted in Figure 73. These resources are especially critical in times of crisis, where individual and family resources may become stretched or constrained; in these cases, it becomes necessary to have a wider range of other options available. We saw in this study that individual or household resources were just not enough to cope, and people soon scrambled to find more 'legs to put under the table' within their local geographies and social networks. The existence of social capital and place-based resources can thus serve to *expand* the options available to an individual by being able to draw upon skills, materials, and facilities outside one's own. This is shaped by the resources and support that are available in one's local context, something that became extremely clear during COVID as people were confined to a specific area.

The context of place has long been acknowledged in shaping the resources available to achieve food security (Mammen, Bauer and Richards, 2009). This can be seen most easily through the research on food deserts, or areas of low food access; these areas are often characterised by having a limited variety of small shops and minimal provision of fresh and healthy foods (Shaffer and Gottlieb, 2002; Baker *et al.*, 2006; Jetter and Cassady, 2006; Shaw, 2006; Hilmers, Hilmers and Dave, 2012). In this context, place determines what types of foods are available to people and at what price; it also determines mental and physical capabilities by affecting access to the foods needed to live a happy and healthy life.

During COVID, we saw that one's 'place' became smaller as people were urged to shop close to the home and avoid public transport; at the same time, the closure of many businesses meant that what was available to people within their local areas also changed. Participants recounted the need to shop at local, more expensive shops because they could no longer access larger, budget supermarkets located farther away, which further constrained financial resources. Some organisations and community spaces also had to close with lockdown, which further narrowed the support available to people in their local area – seen in how Gerald was no longer able access the day centres where he normally went for a hot meal. We see that as personal resources were stretched and participants' sphere of 'place' shrank, they simultaneously lost access to the wide variety of resources that they normally drew upon to achieve food security or cope with food insecurity.

Within this context, participants identified time and time again that what helped the most was people. The social networks and community groups that existed within one's local area became a vital lifeline. This shows the importance of social capital in providing support and building resiliency in times of crisis (Rodriguez-Llanes, Vos and Guha-Sapir, 2013; Aldrich and Meyer, 2015; Long *et al.*, 2022). Being able to share resources, talk to others about struggle, and just have someone to rely on during a stressful time was so important. In this way, we can see that social capital expands the options available to cope, and in doing so, eases the burden.

Personal networks were often used to share responsibilities and skills, expand capabilities, and access facilities that were no longer available to people during COVID (Long *et al.*, 2022). For example, Caroline's household shared responsibilities as she focused on shopping and budgeting while her sons cooked for her, something she physically struggled to do. Outside of the household, neighbours were seen as a particularly important resource (Jones *et al.*, 2020), as lockdown measures often constrained contact with friends and family who lived outside of one's local area. Randy's neighbour helped him overcome a loss in physical capability by doing the shopping for him and his family while they were shielding. Linda's neighbour offered her an internet connection, something she could not afford financially, in exchange for cooked meals. This sharing and cooperative pooling of resources and responsibilities was thus vital for people in getting through a challenging time. We can also see that this completely contrasts the neoliberal mindset of personal responsibility and privatisation of resources, which I argue served only to limit and narrow the options available to people to cope during COVID.

Social capital is also vital as a means of *immediate* support during crisis. This is typified through the people and organisations that can be relied upon when crisis strikes and one is perhaps waiting for other types of consistent support. We saw that participants in this study all identified people that they could call upon in times of need. Past studies on food insecurity and poverty have also shown that friends and family are often relied upon for help with money or food when these resources run out for people (Ghate and Hazel, 2002; Morton *et al.*, 2005; De Marco, Thorburn and Kue, 2009; Bartfeld and Collins, 2017; Tarasuk, Fafard St-Germain and Loopstra, 2020). However, when participants' social networks also became stretched and struggling due to COVID, they came to rely on community organisations like Foodhall. It becomes clear that, even if these social resources are not consistently utilised, their existence and ability to be mobilised in times of crisis constitute an important coping mechanism (Blake, 2019). Indeed, social support has been posited as one of the most important factors driving resilience in the wake of stressful life events (Rodriguez-Llanes, Vos and Guha-Sapir, 2013).

Support networks were also an important *emotional* resource during COVID, helping people cope with the stress and emotional burden that many were feeling and this time (Brown and Reid, 2021; Bu, Mak and Fancourt, 2021). As also evidenced in this study, mental health challenges became more widespread and amplified during lockdown in the UK (Li and Wang, 2020; Pierce *et al.*, 2020; O'Connor *et al.*, 2021; Office for National Statistics, 2021; Daly, Sutin and Robinson, 2022), and in many cases this was linked to loneliness and isolation (White and Van Der Boor, 2020). This provides a likely context for why participants identified that talking with people – even just volunteers – was one of their most positive experiences during COVID. Even participants who felt the most isolated or who were experiencing extreme shame and self-stigmatisation recognised how important it was for their mental health to stay connected with and talk to others about their situation. Past studies have also highlighted how social capital has been seen to contribute to better mental health and wellbeing (Greenblatt, Becerra and Serafetinides, 1982; Putnam, 2000; Li, 2007; Helliwell, 2011). The ways that personal networks contributed to emotional support also helped in strengthening individual resources, such as mental capability. Liam in particular described how talking with his alcohol support worker helped him work through the intense emotions he had around splitting with his partner and having to leave his family home, which helped him better focus on managing new household affairs on his own.

Social networks have also been cited as a vital source of information about support (Aldrich and Meyer, 2015), especially during COVID (Brown and Reid, 2021; Mao, Fernandes-Jesus, *et al.*, 2021). This was seen in this study through how the vast majority of participants learned about Foodhall by word of mouth. As many participants faced extreme barriers in accessing government support or being able to find the help they needed, being able to draw upon social networks to find support was vital. Obviously, this was complicated by lockdown measures and the closure of spaces where one might interact with people outside of their main social sphere. The loss of these 'weak ties' during lockdown has been posited as a major loss of crucial social support (Long *et al.*, 2022), although perhaps overcome by what many saw as an increase in neighbourly support and interaction during this time (Jones *et al.*, 2020; Brown and Reid, 2021; Kavada, 2022).

It is clear that COVID put the vital resources provided by various forms of social capital under pressure. Lockdown measures decreased the opportunities to socially engage with people, and those with limited phone and internet access especially lost out on social interaction, vital information, and many options for support (Watts, 2020). Participants expressed sadness both from a loss of spontaneous conversations and casual encounters, but also from a loss of deeper conversations and connections with friends and family, which was related to the inability to see people physically. The loss of physical interaction perhaps made it more difficult to talk about the struggles that many were facing. This isolation was compounded by the stigma associated with food insecurity that many felt at the time, which furthered feelings of aloneness. The ability to draw upon personal networks for support was further constrained by the fact that many others were also struggling, and people did not want to be an additional burden.

We therefore see how the existence of social capital expands resources, but also how the lack of it narrows the options available for people to cope. Martin *et al.* (2004) found that even if households have similar financial and food resources, those with less social capital are more likely to experience hunger. The Trussell Trust foodbank network has also listed that the lack of informal support networks can be a driver of food insecurity (Sosenko *et al.*, 2019; Bramley *et al.*, 2021). The most isolated are thus the most vulnerable, and often the ones who are left behind. In a crisis context, it was found that individuals who were elderly and isolated were the most likely to die and not be found for days during the 1995 Chicago heat wave (Klinenberg, 2003). Elgar, Stefaniak and Wohl (2020) found that countries which displayed higher levels of economic inequality, as well as countries with lower levels of certain types of social capital, such as civic engagement and confidence in state institutions, actually displayed higher COVID-19 mortality rates. Within this study, participants who experienced the most severe food insecurity, or who felt most helpless about their situation, were also the ones who described the most intense feelings of social isolation. It can thus be seen that individuals and communities that lack social capital are less poised to be able to cope with crisis and disaster situations (Aldrich and Meyer, 2015). However, the onset of crisis can also bring people together, as people form new connections and groups emerge to tackle local issues and provide communal support (Fernandes-Jesus *et al.*, 2021).

Thus, we saw that even as people's physical sphere and the social connections within it narrowed during lockdown, new connections emerged within a hyper-local context (Kavada, 2022). People came together within their communities, neighbourhoods, and cities to provide support responses during COVID that were tailored to local needs, often attempting to 'fill a gap' of support that was not provided by government or local authority services (Fernandes-Jesus *et al.*, 2021), as seen with Foodhall. What was unique to this time was the emergence of locally-based, self-organised 'mutual aid groups', through which volunteers shopped and collected medicine for people and also provided information and emotional support (Tiratelli and Kaye, 2020; Mao, Fernandes-Jesus, *et al.*, 2021). Local responses like these were clearly vital to the participants in this study, who all relied on a grassroots community project (Foodhall) to help them access food. Many participants in this study also mentioned receiving help from neighbours and other volunteer groups; at the same time, many participants also *provided* help to their communities, for example by sewing masks or cooking meals. The desire to engage in reciprocal relationships – seen as the basis for being an active, productive, and contributing member of society (Chan, Stoové and Reidpath, 2008) – was thus clear throughout this study, as many found ways to give back or expressed a desire to do so in the future. Other studies during COVID have also shown how people generously donated time, vehicles, printers, skills, and even money to help their neighbours and local community (Jones *et al.*, 2020; Fernandes-Jesus *et al.*, 2021).

This sharing of resources and cooperation between people underscores the idea of 'mutual aid' that came to the forefront during this time, seen as a direct opposition to the neoliberal-ableist ideals of individualism, autonomy, and privatisation. The concept of 'mutual aid' itself stems from anarchist roots (Kropotkin, 1902), with a focus on solidarity and bringing people together to address shared needs (Spade, 2020). Mutual aid is thus built on a foundation of reciprocity, aiming to erase power notions and social hierarchies between the 'helper' and the 'helped' and instead focus on mutual and collective care – as commonly put, "solidarity not charity" (Firth, 2020; Spade, 2020; Mould *et al.*, 2022).

Internalising these ‘mutual aid’ ideals of togetherness, solidarity, and reciprocity was crucial to overcome the stigma that many participants felt in having to accept help from others during this time. We saw that this stigma was a direct barrier for participants in accessing support, as many waited until there were no other options left before asking for help. The community groups and neighbourhood care that rose to provide support during this time were crucial in alleviating this stigma, as they helped to shift the narrative that the struggles of COVID were a personal problem to seeing it as a collective community problem. Friendly volunteers were consistently cited as helping participants feel comfortable asking for support and overcoming the intense shame and embarrassment that many felt when first reaching out for help. This was perhaps also aided by the fact that many of the community responses that emerged during this time, like Foodhall and the mutual aid groups, did not ‘means test’ or require proof of need; this meant that help was openly offered without having to prove any state of destitution. The lack of these eligibility requirements likely eased stigma, as these requirements in and of themselves can be seen as categorising people into different boxes outside of ‘normal society’ (Reisman, 2004). We also saw that participants felt better knowing that they were not the only ones struggling during this time, perhaps realising that their struggle was indeed not a personal failure and that they were not ‘in this alone.’

We thus see that a “hyperlocal infrastructure of care” emerged in the face of the global pandemic (Kavada, 2022), created within an intersection of social capital and place. Self-organisation and volunteering in communities were facilitated by a shared social identity, trust, and a sense of belonging (Fernandes-Jesus *et al.*, 2021; Mao, Fernandes-Jesus, *et al.*, 2021; Wakefield, Bowe and Kellezi, 2022), which in the past have also been seen to enable cooperation, collective support, and solidarity within communities (Drury *et al.*, 2019). Thus, having a connection to the *place* where one lives and the *people* there were key determinants of self-organising and therefore of the support that became locally available. These connections did not necessarily have to exist strongly before, as many formed and were strengthened during COVID as people ‘banded together.’ However, strong social capital, existence of local resources, and the creation of local partnerships did help to sustain these volunteer groups, highlighting the importance of cooperation and collaboration (Fernandes-Jesus *et al.,* 2021). Further, participating in these volunteer groups and perceiving an increase in community connection were associated with better mental health and wellbeing during COVID (White and Van Der Boor, 2020; Fernandes-Jesus *et al.*, 2021; Mao, Drury, *et al.*, 2021). Together, this provides further support for Helliwell, Huang and Wang (2014)’s assertion that communities with better social capital respond to crisis more happily and effectively.

5.3.2 Building resiliency through community food infrastructure

Shocks and stresses to the food system are predicted to increase in frequency and severity in the future, especially with the ensuing impacts of climate change (FAO *et al.*, 2022). Many across the UK are still struggling with the inadequacies in the state welfare system and have been unable to recover from the hardships brought by COVID (Bramley *et al.*, 2021). At the time of writing, the UK is already experiencing another crisis seen with a dramatic rise in the cost-of-living, spurred largely by supply chain disruptions following Russia’s invasion of Ukraine in 2022 (Webster and Neal, 2022). Drastic increases in fuel and food costs are plunging people into states of immense hardship (Goodwin, 2022; Khan, 2022), leaving many in situations where they must decide if they will "heat or eat" during the winter months (White, 2021).

This points to a pressing need to build social capital, local partnerships, and place-based connections in order to foster community resilience in the future. Community organisations and spaces like Foodhall are crucial to building resilience into the food system, and thus these spaces should be developed and supported as part of a community's food infrastructure. These spaces can rapidly adapt to provide fast-acting crisis response – seen poignantly through the way that Foodhall created a new emergency food provision service during the first week of the COVID-19 lockdown (Koseda, 2020). They serve as key spaces for people to build the skills, capacities, and social relationships that can be drawn upon in times of need. This goes further than the traditional charity role of 'filling a welfare gap' by building solidarity within the community, bringing forth the ideals of mutual aid and making support more accessible through reciprocal care. This also provides a platform for grassroots social action to challenge and transform the dominant neoliberalist individualism that separates and isolates people during the times when they need each other most.

Foodhall was only one of many community projects that rapidly adapted during COVID to meet the needs of their local communities. Other kitchens and social eating spaces, such as those in the UK-wide National Food Service network, similarly shifted their physical spaces and developed new structures to deliver food and essential goods to people rapidly and effectively (NFS, 2021a). Other local food actors, such as local farms, also stepped up to 'fill the gap' in government programmes and a crumbling national food supply chain (Jones, Krzywoszynska and D. Maye, 2022). This research showed that these types of community responses were a lifeline to people during COVID, in some cases being the only support that people could actually access and truly rely upon.

An essential resource for community efforts like Foodhall was the social capital and connections built in these spaces prior to COVID (Fernandes-Jesus *et al.*, 2021; Jones, Krzywoszynska and Damian Maye, 2022). This is what allowed these groups to come together and mobilise to meet community needs during a time of crisis. In Foodhall's case, this was seen through the number of volunteers that the project was able to draw upon during COVID – using over 500 volunteer hours a week to distribute food at some points – as well as the sharing of resources that occurred with other food projects both inside and outside of Sheffield. For example, branch projects of the National Food Service consistently met (virtually) during the pandemic to share best practices and successes around new delivery systems and COVID-19 safety policies. This shows the importance of developing social capital and fostering partnerships and connections to enable resilience to crisis.

Food insecurity and financial struggle often serve as barriers to engaging in social life (Runnels, Kristjansson and Calhoun, 2011; Purdam, Garratt and Esmail, 2016), thus threatening the ability of people to form and maintain the social networks that serve as critical support systems during times of crisis (Blake, 2019). Community spaces have been seen to play a crucial role in fostering social capital and supporting health and welfare (Oldenburg, 1989; Putnam, 2000; Cattell *et al.*, 2008; Peoples, 2015). Outside times of lockdown, social eating spaces and 'contribute-what-you-can' cafés like Foodhall can provide the space and opportunity to meet new people and have a hot meal in an unstigmatised setting; this means that the experience of going out and having a coffee or meal with friends is not isolated to just those who can afford it.

The importance of social eating spaces specifically in building social capital is underscored by the crucial role that food itself plays in this process, serving as a “social glue” and common unifier between people (Blake, 2021). Food is much more than just a source of nutrition or way to meet basic physiological needs; it has symbolic and cultural meaning and is linked to memories, personal experiences, and personal identity (Dowler *et al.*, 2009; Arvela, 2013). The action of eating together facilitates social bonding, providing a context to form new connections and also deepen existing ones (Fischler, 2011). Evolutionary psychologist Robin Dunbar even suggests that the act of eating socially may have evolved as a mechanism to facilitate the formation of social bonds – bonds which are associated with better health, wellbeing, and survival outcomes for people (Dunbar, 2017).

The inability to participate in social eating can thus be associated with feelings of isolation and loneliness (Bofill, 2004; Björnwall *et al.*, 2021), reducing the critical social resources available to people (Dunbar, 2017). In this study, the loss of being able to choose or access certain foods was linked to feelings of lost control and personal identity, as well as lost connections with people. Thus, we see that there are complex ties between social and food-related experiences. Creating accessible spaces where people can come together around food can be an important and potentially overlooked strategy toward building resilience, both on individual and community levels.

Research has shown that a *diverse* and *wide* array of social capital enables resilience (Jones, Krzywoszynska and D. Maye, 2022). Social eating spaces provide the opportunity to build a diversity of social resources by facilitating the development of several types of social capital. For example, Foodhall encourages everyone who visits the project to be involved and take ownership (‘civic participation’), thus contributing to building the sense of belonging and social trust that is critical for self-organised community response efforts (Mao, Fernandes-Jesus, *et al.*, 2021). In addition, accessible and open community spaces like Foodhall allow people from a wide range of backgrounds to meet (i.e., ‘bridging social capital’), which can provide opportunities and support outside one’s normal social sphere (Granovetter, 1973). Places like Foodhall can also bring together people who might be experiencing similar life challenges, providing a space to talk through struggles and share information about support options. Indeed, many participants in this study found it hard to share about struggle with friends and family, but were able to share more openly with support workers and others less close to them. Finally, consistent engagement in community projects can also make existing friendships stronger (‘bonding social capital’), thus contributing to the number of people that one can rely upon in times of need (Fernandes-Jesus *et al.*, 2021).

Reciprocity underscores the formation of social relationships, and thus is a critical part of human life (Sen, 1999a; Becker, 2005; Maiter *et al.*, 2008). Spaces like Foodhall aim to facilitate these exchanges of care between people, which, as seen in this study, is crucial to overcoming the feelings of stigma that stem from accepting support. Providing options for reciprocity helps erase the power hierarchies that often exist between the ‘giver’ and ‘receiver’ of food support (Marovelli, 2019). This allows for mutual aid and collective care to emerge, with this notion of ‘solidarity not charity’ directly opposing the neoliberal individualism that left many in this study weighed down under an immense mental burden.

We can thus see that Foodhall not only provided logistical support in terms of food for people, but also provided an opportunity to build the capabilities and connections that increase resiliency to crisis in the future. These necessary and multiple ‘rungs of support’ are presented within Blake (2021)’s ‘Food Ladders’ framework, which identifies the need for catching, capacity building, and transforming strategies to build community resilience to food insecurity. In Foodhall’s case, these rungs are seen through the ‘catching’ logistical support they provide with food deliveries, the ways that they help to build capabilities through supporting health initiatives, volunteering, and skill sharing, and the work that they undertake with the National Food Service to transform the structures that shape the injustices and inequalities in the food system. This holistic support provides the space for people to address the multiplicity of challenges tied to food insecurity whilst logistical burdens are eased (Blake, 2019).

Community food projects serve as critical intermediaries within a local food system, connecting with food producers and shops to redistribute surplus and also forming key connections with other community and health support services. Local food actors have often faced barriers in providing food to traditional charities (such as food banks) because of specific requirements and difficulties in utilising smaller quantities of fresh produce (Krzywoszynska, Jones and Maye, 2022). This points to a need for more small, flexible, and grassroots community kitchens like Foodhall that can (and do) process and redistribute these foods to those who need it. Building connections between local farmers, producers and community groups helps to ensure that all people can access good quality food (Nelson and Stroink, 2014).

This lends to the argument that communities are suitably positioned to undertake crisis responses to ensure that specific place-based needs are met and that no one is left behind. Indeed, the National Food Service members stipulate that community food projects are well-poised to lead local crisis responses, as these groups are aware of the specific needs of their communities and are already well-connected to each other, as well as to other community projects, food system actors, and local authorities (NFS, 2021b). We saw how crucial these local collaborations were in providing immediate support for many of our participants, who had exhausted many other options and were often left behind by other government support programmes.

However, the immense amount of pressure placed on an under-resourced voluntary sector during COVID shows the unsustainability of this approach in the current social and political climate. If community groups are to contribute to crisis response, it is also imperative they have resources and funding to ensure reliability, consistency, and sustainability in these efforts. Many local businesses and community organisations were unable to recover from the financial hit of COVID – including Foodhall itself. After seven years, Foodhall closed at the end of 2022 due to a lack of funding. Although this organisation was resilient in the face of the COVID crisis, the project could not sustain the financial losses from COVID. This was largely spurred by the sizeable expenses the project undertook during COVID to buy food and essentials for people; although much food at the time was donated by local businesses and even the city council, these donations were not enough to meet the extreme demand for emergency support.

It thus becomes clear that more support is needed for these community spaces (NFS, 2021b), which should include funding and material resources, but could also include allocations of physical buildings to make these spaces a critical and permanent part of a locality's food infrastructure. Support should be given to community groups in ways that preserve the integrity of the group and do not draw away from the role of communities in decision-making processes (Fernandes-Jesus *et al.*, 2021). Local authorities should work closely with community groups to draw upon their vast knowledge of local needs, working together to implement collaborative solutions and to provide consistent, equitable, and reliable access to food in ways that are culturally appropriate and socially engaging.

Recognising a right to food within UK law could be an important step toward the development of necessary food infrastructure (De Schutter, 2014). As social eating is a common human experience that underscores living a happy and healthy life (Dunbar, 2017), it is important to have spaces where everyone can access this experience, regardless of financial status. Building on the idea of a 'National Food Service', this community food infrastructure can thus serve as a critical place-based resource that also builds individual capabilities and social capital, ensuring that healthy, affordable, and appropriate food is available, accessible, and utilisable by all.

5.4 Chapter conclusion

Drawing on the narratives of fourteen individuals facing barriers to accessing food during COVID, we uncover the complex and multidimensional nature of food insecurity. We find that the ability to access and utilise the food one needs to live a healthy life is not dependent on only financial resources, but also requires a range of other individual, household, place-based, and social resources that must be consistently available. COVID diminished a suite of these resources all at once, as many were in financial crisis, isolated from friends and family, and struggling to cope with physical and mental illnesses, addiction, trauma, and newfound stress, all whilst support options became more difficult to access under a government-imposed lockdown. It thus becomes clear that addressing food insecurity requires addressing the multiplicity of hardships that are linked to, coincide with, and exacerbate this experience. This shows a need for more holistic support that eases logistical challenges, includes mental health care, and contributes to positive physical health, thus providing the options and mental space needed to build capabilities and plan ahead.

As individual resources were constrained or disappeared entirely during the pandemic, and an embodied neoliberal-ableism further individualised and intensified this burden, what the participants in this study relied upon were the key social resources within their communities. This social capital was seen through the personal networks and the many community groups that were created or adapted during this time. When the market failed to provide a consistent and affordable supply of food, and when so many were falling through the cracks of government programmes, we saw immense resilience in local communities as they mobilised to provide reliable, flexible, and fast-acting support to those around them. This shows a need to address key inadequacies in the state welfare and care systems, whilst also continuing to develop and maintain the community-based infrastructure that allows for people to come together and share the load during crisis. These community spaces are critical to building the skills, capacities, and social relationships that one can draw upon in times of need. However, there must also be adequate support to ensure the reliability and consistency of these spaces.

This is particularly poignant as Foodhall, which served as a key source of support for thousands across Sheffield during the pandemic, has now had to close due to the financial burden sustained during this time. The drastic loss of this crucial community space further demonstrates the need to ensure that these spaces remain as critical parts of a local food infrastructure. Foodhall was a lifeline during COVID, but it was also a space that became a home for many in the Sheffield community. Although the doors are now closed, the connections made through the space and the momentum built to challenge the injustices within the current food system have not been lost. The concept of Foodhall lives on in the friendships made, the recipes shared, the conversations and ideas that took ground, and the plates of food passed between hands.

Chapter 6: General discussion and conclusion

This thesis explores various concepts of local food system sustainability and resiliency using three interdisciplinary method frameworks, as depicted in Figure 74. First, lifecycle assessment (LCA) was used to identify the main impacts from crop production, processing, and transport on different types of local farms in the U.S. and UK. Then, soil health and ecosystem service assessments were performed on a subset of these farms to provide additional context to the other services or disservices that come from the farm as a whole. Finally, a qualitative methodology was applied to elucidate the major barriers to achieving food security and related health and social challenges during a time of crisis, thus providing insight to key aspects of wellbeing. As these methods build upon each other down the food supply chain, from crop production to the consumer, this facilitates a more holistic evaluation of sustainability using a food systems perspective. Further, as these methods are combined and outcomes are fed back to each other throughout the food supply chain, this provides opportunities to build resiliency into the system. In the following sections, these three study frameworks (Figure 74) and major research outcomes are collectively explored, highlighting the opportunities and challenges for local food systems to contribute to sustainability.

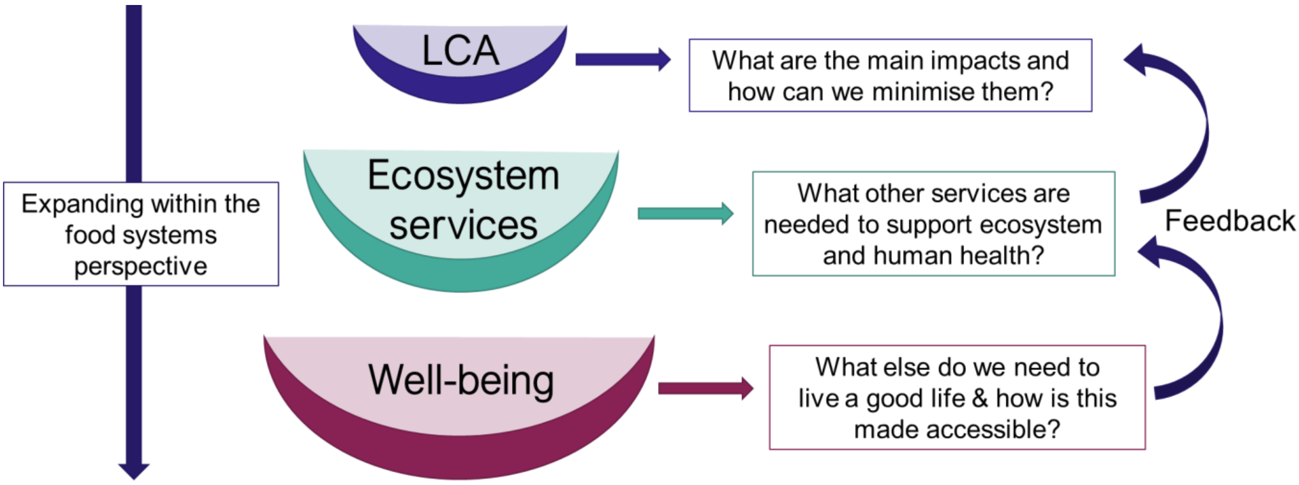

Figure 74 – *Main aspects and methodologies for food systems explored in this thesis, including: lifecycle assessment (LCA), ecosystem services, and well-being.*

6.1 Summary of methods and outcomes from a food systems perspective

LCA was the first method employed in this thesis (Chapter 3), used to compare different models of local agriculture and test the common assumptions that 'organic' or 'more local' produce is more environmentally sustainable. LCA is a robust accounting framework that allows for resource use and associated emissions to be tracked and summed throughout a product lifecycle, in this case for the crop; this assessment thus provides key insight to resource use efficiency in relation to product output (Garnett, 2014). LCA has therefore been highlighted as a central methodology for evaluations of food system sustainability, as long as detailed information on inputs throughout the supply chain can be obtained (Horton *et al.*, 2017). The benefit of this methodology within this thesis is that it provided a rigorous quantitative analysis of local food supply chains, uncovering the materials and processes that contributed most to certain environmental impacts. For example, it was found that compost use was the main determining factor influencing impacts on most organic farms, which has not been widely identified in previous research (Perrin, Basset-Mens and Gabrielle, 2014; Dorr *et al.*, 2021). The method is also key from a food systems perspective because trade-offs between different types of impacts, such as global warming, eutrophication, acidification, and land and water use, can be identified; this is critical to prevent unintended consequences from changes in land management. In this thesis, we saw that organic farms tended to have higher land use impacts, even when other impacts were relatively low; this was especially seen for farms using cover crops as a main fertility input. This outcome is important to consider when deciding whether agricultural land should be managed for land sharing or land sparing.

Additionally, the use of LCA to evaluate individual farm case studies, using a highly granular dataset provided by farmers, gave key insights to specific management practices that drove variation between farms. Most LCA studies focusing on farm case studies generally have a much lower number of participating farms (usually no more than 5-6, although more practically 2-3); this study is the first (to the author's knowledge) to evaluate such a wide array of local farm case studies all together, including 25 farms and 40 crop lifecycles across two countries. Many other studies use industry-averaged data, which can also dilute the variation between individual farms within a particular category (e.g., organic vs. conventional, or urban vs. rural). When comparing farm category averages in this study (using the default allocation method), the lowest average impacts were observed for the rural conventional category in the U.S. and the rural organic category in the UK. However, when evaluating farms individually, certain peri-urban organic farms in both countries had similar and sometimes even lower impacts than farms in the lowest-impacting farm categories. At the same time, the peri-urban farm category also included some of the highest impacting farms. Thus, what this study clearly highlights is that individual management practices on the farm, more than anything else, determine impacts resulting from food production. Additionally, because most impacts from seedling production through distribution come from the farm stage, this means that changes in farm management present a key opportunity toward increasing environmental sustainability within the supply chain.

Although the rigorous, quantitative framework of LCAs clearly has the potential to uncover key impact drivers on farms, there are also disadvantages to this method. In particular, LCAs are very subjective to methodological decisions made by the practitioner, and at this time there is no standardised set of methods used consistently within agricultural or food LCAs (Arzoumanidis *et al.*, 2013; Notarnicola *et al.*, 2015). The impact of differing LCA methodologies or assumptions was clearly seen in this thesis through the application of different allocation scenarios. The impact trends between farm categories could basically be reversed by changing the allocation assumptions applied to just one component of organic farm inputs – compost. For example, relatively high global warming, fine particulate matter formation, terrestrial acidification, and marine eutrophication impacts were observed on more urbanised farms in allocation Scenario 1, where compost production for inputs was included. This scenario thus suggests an inefficient use of resources per crop output on urbanised farms, with the obvious conclusion being that these are not ideal farm models. However, when viewing compost as a waste management strategy and therefore excluding production burdens for inputs (allocation Scenarios 2 and 3), these higher impacts were largely diminished; when considering the avoided burdens from alternative waste streams, impact trends for global warming and marine eutrophication were in fact reversed. In these scenarios, composting is now being valued as a method of urban waste cycling, and urban farms are seen as a key outlet for municipal compost waste streams, thus potentially contributing to the development of city-wide circular economy models (Rufí-Salís *et al.*, 2020). If this is a key aim for a particular city, it would seem that viewing urban farms as 'waste cyclers' and allocating benefits for compost use would be more appropriate.

The multiple functions desired from a specific food or farming system should therefore be considered to inform both the scope of an LCA and the assumptions and methods applied. Another option, particularly relevant from a policy perspective, could be using the consequential LCA approach. In contrast to the attributional LCA method used within this thesis, which captures the current resource flows and emissions in relation to a specific output, consequential LCAs examine various consequences that result from changes in product output (Finnveden *et al.*, 2009). This could be helpful to evaluate interconnected food systems aims and identify indirect or 'knock-on' effects on other systems outside the crop lifecycle (Meier *et al.*, 2015).

Despite the overarching method used, it is impossible to capture every trade-off or environmental service within an LCA. Within a food systems perspective, the use of LCA can be quite narrow because it tends to focus on one output, rather than considering the multiple functions that can be provided by farming systems (Bernard and Lux, 2017; Boone *et al.*, 2019). This has led to the critique that LCA lends itself toward an efficiency-oriented food systems perspective (Garnett, 2014), thus disadvantaging agro-ecological and/or organic farming systems, which tend to trade off the ability to maximise food production to also provide a suite of other environmental and/or social services. Thus, the use of LCA is better seen as an evaluation of environmentally-efficient production, rather than as a full sustainability assessment (Garnett, 2014; Boone *et al.*, 2019; van der Werf, Knudsen and Cederberg, 2020).

Taking this into consideration in Chapter 4, the LCA foundation was built upon to evaluate other ecosystem services that were not captured in this framework – i.e., to see 'what else was missing' (Figure 74). The focus was placed on evaluating soil health, as soils are the supporting basis from which the majority of land-based ecosystem services flow (Dominati, Patterson and Mackay, 2010; Robinson *et al.*, 2013; Gomiero, 2016). Soil health was generally observed to be better in organically-farmed and uncultivated spaces on farms, which points to increased ecosystem service provisioning from these soils, such as better flood mitigation, improved soil structure, erosion control, and carbon storage. Without evaluating soil health, the benefits from many of the organic farms (especially the urban ones) would have been lost, as the LCA study showed many of these farms to be largely inefficient from a food production standpoint, although in reality they are actually co-producing a range of other services. Additionally, as uncultivated spaces on farms are 'unproductive' from a food standpoint, these spaces are generally not accounted for within LCA frameworks (Boone *et al.*, 2019); thus, without expanding the study to evaluate these spaces on farms, a key source of ecosystem service provisioning would have been ignored.

While there is much onus placed on finding ways to increase food production in an environmentally sustainable way to ensure future food security, other critical aspects are often lost in this agricultural research space, particularly the factors that influence food security outside of food supply (Sen, 1981; Holt-Giménez *et al.*, 2012; Holt-Giménez and Altieri, 2013; Tomlinson, 2013). Being food secure is a key component of human well-being, or living a happy and healthy life; thus, food security is interrelated to other determinants and components of wellbeing, such as access to basic materials and resources for a good life, health, and good social relations (Millennium Ecosystem Assessment, 2003). Wellbeing frameworks are linked to and build upon the idea of ecosystem services, as these services provide the foundation to support human life (Millennium Ecosystem Assessment, 2003; Powlson *et al.*, 2011). However, as these services and related resources are not universally distributed or universally accessible to everyone, a key part of ensuring global food security for *all* is also seeking *who* is left out and *why*. Even in cities and countries where food is seen to be widely available, many still experience food insecurity (Holt-Giménez *et al.*, 2012); thus, a key question becomes 'what else is needed?' (Figure 74). To uncover these issues, qualitative methodologies are ideal as they can provide rich insight to specific barriers and challenges to achieving food security within local contexts (Hall and Holmes, 2020).

In Chapter 5, issues of food insecurity were explored during a time of crisis, drawing on the narratives of those experiencing barriers to accessing food during COVID-19 in Sheffield, England. Although disruptions in the food supply during this time made it difficult to find affordable foods in one's local area, this research showed that food insecurity was also spurred by a multitude of challenges that diminished the individual, household, social, and place-based resources necessary to achieve food security. A consistent food supply, albeit crucial, is only one aspect of food security; ensuring that people can physically access food, afford it, cook it, and eat it requires expanding evaluations to consider geographical, socio-economic, and political contexts (Sen, 1999a). One must also acknowledge how food insecurity is linked to poverty, social isolation, and physical and mental health challenges, which are not necessarily considered as a direct aspect of the food system. However, as discussed in Chapter 5 (Figure 73), the ability to achieve food security is largely dependent on a wide range of other resources and local networks that can simultaneously address these challenges. Thus, this research helped to identify important elements of resilience within a community's food infrastructure.

It is essential that the outcomes of these different methodologies and perspectives feedback to each other to inform future research; indeed, one of the largest challenges in food systems research is its complexity and interdisciplinarity, which can lead to different aspects of the system being siloed within academic disciplines or different policies. When this happens, trade-offs between aims or functions of the food system may be lost (Zurek *et al.*, 2018), leading to unintended and potentially damaging consequences. For example, this was seen within the first Green Revolution; although the aim was to increase food production to decrease hunger, which happened, it came at a cost to many local environments and traditional smallholder livelihoods (Cullather, 2004; Harwood, 2011; John and Babu, 2021). Thus, the challenge for scientists and researchers today is to 'open out' our work and perspectives, finding ways to facilitate communication across disciplines, and with industry and policy-makers, to capture and address the complexities of the food system, rather than reducing the scope (Carpenter *et al.*, 2009; Powlson *et al.*, 2011; Norton *et al.*, 2016). It is crucial for researchers to gain the trust of the communities that we work with, and particularly to foreground the voices of marginalised or underrepresented groups to provide context for what is actually needed to meet the food system aim of food security for *all*.

6.2 Key insights, challenges and opportunities for local food systems

Considering the many complex issues tangled within food and agriculture, ensuring the sustainability of this system is one of the most pressing and complicated global challenges. Food pervades all parts of human life – it is necessary for survival, but is also a key source of pleasure, a symbol of culture, an avenue to social life, and an embodiment of care (Fischler, 2011; Hanssen and Kuven, 2016; Dunbar, 2017). Due to the many complexities of food and its connection to basically all realms of society, food system sustainability can be seen as one of the main wicked problems of our time. ‘Wicked’ problems are seemingly impossible to solve due to the interconnected issues and the many conflicting interests that define them (Batie, 2008; Peters and Pierre, 2014). This results in a problem that has a seemingly infinite series of trade-offs with no one answer. Unfortunately, the work of this thesis only concurs with this sentiment – there are a wide range of potential solutions or strategies for food system sustainability, but their application must depend upon local context; there is no universally-accepted ‘one way forward’ for food system design.

The scope of this research focused on the potential for local food systems to contribute to food system sustainability and resiliency. The outcomes showed that ‘more local’ food production is not always more environmentally sustainable, when viewing this based on resource use efficiency. However, integration of organic farms in and around cities may aid in soil remediation and provide a range of other ecosystem services. Additionally, more local control of food systems that facilitate connections between food system actors can provide flexibility and enable resiliency in times of crisis. Thus, the key opportunity for small-scale local farms is their ability to provide ecosystem services and build resiliency within their communities; the key challenge is improving efficiency within these models.

The following sections provide further detail on the key insights, opportunities, and challenges uncovered from the studies in this thesis and from the informal conversations had with farmers, food producers, food charity workers, academics, artists, and local and national government officials throughout the four years of this research.

6.2.1 Division in the agri-food sector

The agri-food industry and academic sector is characterised by stark divides in the practices that are promoted to increase food system sustainability (Shennan *et al.*, 2017). This is seen plainly with the land sparing vs. sharing arguments (Green *et al.*, 2005; Phalan *et al.*, 2011; Bernard and Lux, 2017), and also with the divisive views of organic vs. conventional farming (Rosen, 2010; Rodman *et al.*, 2014; Pew Research Center, 2016; Koswatta *et al.*, 2023). In extreme perspectives, conventional farms are vilified as industrial, extractive regimes (Beus and Dunlap, 1990), and organic foods are either for ‘hippies’ (Sligh and Cierpka, 2007; O’Sullivan, 2015), or, more recently, are just “luxuries for the rich” that cannot significantly contribute to global food security (Bai, Wang and Gong, 2019). This divide is spurred by the evaluation of agricultural strategies as universal or as a ‘a one size fits all’ approach. This is seen in research that focuses on questions such as: “can organic farming feed the world?” (Badgley and Perfecto, 2007; Kirchmann *et al.*, 2009), which indeed seems a valid and interesting assessment, but consistently produces answers on both sides and in actuality is not particularly useful considering that the current conventional food system is also not feeding the world and that this depends on a range of other socio-political factors (Holt-Giménez *et al.*, 2012). A more appropriate question for organic and conventional management would be: "in what local contexts do these management practices make more sense to better service the needs of the local environment and the community whilst also contributing to global ecosystem and human health?"

Therefore, although 'organic' vs. 'conventional' agriculture is an easy and useful nomenclature to classify a set of farming practices, these classifications lose out on specific context, with the actual differences between individual farms not being so clear-cut. This author argues that these classifications serve to target and divide growers based on a certification label that, in the end, does not provide an all-encompassing measure of what it means for a farm to be socially, environmentally, and economically sustainable. In reality, organic and conventional farms exist along a gradient of what is seen as more 'industrial' or more 'ecological' (Shennan *et al.*, 2017). This was especially exemplified throughout the LCA studies in Chapter 3, which saw drastic variation in impacts between individual organic farms, and where outcomes for both organic and conventional farms challenged pre-held notions.

Through the LCAs, we saw that many organic farms in both countries were able to achieve yields higher than conventional farms (excluding UK tomato production), some even with lower environmental impacts; this challenges previous research that tends to assume that organic farming results in lower yields or will have higher environmental impacts when evaluated based on product output (De Ponti, Rijk and Van Ittersum, 2012; Seufert, Ramankutty and Foley, 2012; Tuomisto *et al.*, 2012; Meier *et al.*, 2015). This also points to the importance of small farms as provisioners for local communities (Laughton, 2017). The soil health assessment further showed that these farms were able to contribute to increased ecosystem service provisioning. At the same time, certain organic farms were also characterised by inefficient food production and high resource use. Although not necessarily seen in this study, it is also important to note that not all organic farms are the multifunctional, diverse systems that we tend to imagine (Kremen, Iles and Bacon, 2012). Although this is not conducive to the values of organic farming (FAO, 1999; IFOAM, 2020), there are industrial, monoculture organic farming systems that also exist (Kremen, Iles and Bacon, 2012; Tscharntke *et al.*, 2021).

The term 'conventional' farming tends to be more aligned with the idea of unsustainable, intensive monoculture production (Beus and Dunlap, 1990), but actually holds little value other than the idea of 'non-organic.' In actuality, the resource efficiencies and economies of scale seen on conventional farms can lend toward lower environmental impacts, seen particularly in the case of the rural conventional farms in Georgia, USA. Additionally, many conventional farmers can (and do) employ practices that provide ecosystem services, which are often associated more with organic farming. For example, although signs of soil degradation were observed on some UK conventional farms through the research in Chapter 4, this was not universal; one conventional farm in particular had soil health characteristics more similar to the organic farms and was employing environmentally-friendly management techniques such as cover cropping and minimum tillage. Thus, we see that an isolating view of conventional farmers as exploiters of the land can erase their actual role as land stewards.

6.2.2 A range of solutions for local contexts

This division within the agri-food industry and academic sector can detract from the investigation and promotion of the specific practices and system transformations that are needed for a sustainable food system. It becomes clear that, rather than universally advocating for one agricultural approach over another, what we actually need is a combination of the best farm models, and best farm practices, for a given local context to support the multifunctional aims of the food system. Employing a diverse set of farming models builds resiliency within a local food system and allows for these different models to service specific community and ecosystem needs (Hendrickson, 2020).

A key outcome of this research is the importance of local context when evaluating strategies for food system sustainability, both in terms of the geographical, social, political, and economic sphere within which a farm sits, and the specific management practices employed on the farm. For example, increasing local food production does not make sense for every geography, or for every crop, seen with the classic example of domestically-produced glasshouse tomatoes in the UK being higher-impacting than those imported from Spain (Williams *et al.*, 2009). Within this research, the ideal model and scale of local agriculture also varied depending on the country investigated; rural conventional farms generally resulted in the lowest average impacts in the U.S., whilst rural organic farms resulted in the lowest in the UK.

However, using LCAs to make blanket statements about farm sustainability is not recommended; the soil health assessment performed in the UK showed that organic farms in particular likely contribute to a range of other ecosystem services not captured in LCAs. These complexities when investigating food system sustainability prove particularly challenging for policy-makers who are aiming to make large-scale land / resource management decisions or support 'sustainable practices' at a national level (Norton *et al.*, 2016). In this case, it is essential to apply a food systems perspective to consider trade-offs within the set of goals that a specific community, or population, wants from their food system, while also recognising the potential for unintended consequences in other places (Zurek *et al.*, 2018).

In this thesis, we found that rural organic farms in the UK had the lowest environmental impacts in the LCAs, good soil health, and lower heavy metal contents than other urbanised farms, which suggests efficient food production and ecosystem service provisioning. This would imply that this farm model is the ideal, meeting needs of (relatively) high food production, low resource use, and high ecosystem service outcomes. However, there are some impracticalities of expanding this farm model within the UK context. Increasing the matrix of relatively small-scale, labour-intensive organic farms in the UK would require a suite of young people who are willing and able to pursue this as a business venture, which is currently seen as a key challenge in UK farming (Shields, 2019; FarmingUK, 2022). Land access is considered as one of the main obstacles for new entrant farmers, although national programmes and policies are currently being developed to support new entrants in this regard (Defra, 2020). Thus, it is clear that encouraging more small-scale, rural organic farms must also come with societal and policy changes that make this a viable professional route for young people.

As the current UK farming system is not dominated by these small-scale farming models, it would perhaps be most advantageous to support the widespread adoption of the practices that were seen to contribute to lower environmental impacts and better soil health on the UK rural organic farms, particularly high organic matter incorporation, use of cover crops, and minimal machinery use. Indeed, the prior two of these are actually being supported within England's new Sustainable Farming Incentive scheme (Defra & Rural Payments Agency, 2022c). By promoting these sustainable practices across all farms (which are often part of organic farming systems) and also facilitating the start-up of new, small-scale organic farming projects, this diverse set of farms can provide a varied range of public goods that include more than just food. Indeed, this is a key goal of the UK's new agricultural policies (Defra, 2020). The need to incorporate a range of sustainable farming models to serve the multifunctional aims of the English food system was also expressed by Henry Dimbleby in the National Food Strategy for England report (Dimbleby, 2021). The report particularly highlighted the need for a combination of land sparing and sharing strategies to provide multiple ecosystem services (e.g., biodiversity and carbon storage) and cater to multiple species (some of which thrive on diverse farmland) (Maskell *et al.*, 2013; Finch *et al.*, 2019).

For Georgia, USA, the highest yields and lowest average impacts were generally seen on the rural conventional farms (using the default allocation scenario). This suggests that these farms are employing a resource-efficient production system, which lends itself toward an argument for sustainable intensification and land sparing. Of course, the major concern with blanketly promoting the land sparing strategy is whether land will actually be 'set aside' for conservation purposes, or instead used for further intensive agricultural production to increase profits (Fischer *et al.*, 2011; Desquilbet, Dorin and Couvet, 2017). The promotion of this strategy would thus require large-scale land-use planning and zoning for conservation that would need to be supported by farmers (Desquilbet, Dorin and Couvet, 2017).

The research in Georgia also showed that certain peri-urban farms achieved similar or sometimes even lower impacts than the conventional farms. Unlike in England, rural organic farms in Georgia did not emerge as an ideal model. These farms (along with certain peri-urban farms) saw relatively high transport impacts due to the inefficient transport of relatively small amounts of produce over longer distances than was generally necessary for UK rural organic farms, as England is characterised by a denser fabric of urban areas. The ideal locations of organic farms in Georgia were therefore those closer to final markets of sale in urban and suburban areas, which also provided access to a larger labour pool, an aspect that was a challenge for rural organic farms. In particular, the peri-urban farms that sold most of their produce on site or at nearby farmers' markets, used a moderate amount of organic fertiliser inputs (producing some on the farm), and achieved moderate to high yields emerged as the ideal model for small-scale organic farming in Georgia. Although soil health was not tested on the farms in Georgia, the research from the UK points to the likely fact that these farms are also contributing to improved ecosystem service provisioning in comparison to the conventional farms. We thus see that, for Georgia, a combination of efficient conventional production with small-scale organic farms located on urban fringes seems ideal to meet food and ecosystem service needs, although management practices that build soil health should be promoted throughout.

Across both countries, generally high impacts were observed on urban farms, which was mainly (but not exclusively) a result of high resource use (compost). This implies that urban agriculture, as the epitome of ‘hyper-local’ production, may actually result in highly inefficient use of resources and may not be the ‘sustainable solution’ previously espoused in literature (Lovell, 2010; Pearson, Pearson and Pearson, 2010; Specht *et al.*, 2014; Thornbush, 2015). However, this author would argue that the context where these urban farms are located, why they exist, and what else they are doing (besides producing food) is also important. For example, although the LCA identified high impacts on urban farms that were mostly related to high compost use, the soil health assessment on UK farms uncovered that the application of compost may also be contributing toward the healthy soils seen on these urbanised farms. This suggests that these farms may actually be contributing to urban soil remediation, as the structural degradation issues commonly associated with urban soils were not seen (e.g., compaction, low soil carbon) (Pouyat *et al.*, 2010; Beniston, Lal and Mercer, 2016). Thus, encouraging organic farming in cities may also be an avenue toward improved land stewardship. Without employing the interdisciplinary, multi-method approach used in this thesis, this additional context may have been lost.

Additionally, urban farms may be important for addressing specific social challenges within a city, which were not uncovered through the LCA or the soil health assessment, further showing a need for interdisciplinary food system research. The city of Atlanta, GA provides an optimal example. This city has historically been characterised by problems of food access, particularly cited as having the third highest density of food deserts (i.e., low-income, low-access census tracts) out of all U.S. cities in 2014 (Burns, 2014). At this time, approximately half of all Atlanta residents were living in a food desert area, which comprised 59% of the city’s land area; these areas were concentrated mainly in the south of the city, which is characterised by a majority of low-income, black neighbourhoods in comparison to higher-income, mainly white neighbourhoods in the north (Ross, 2014; City of Atlanta, 2020). This massive inequity in food access, spurred and underscored by the structural racism that defines the city’s structure (Ross, 2014), led to city-wide goals of increasing access to fresh foods (City of Atlanta, 2020; City of Atlanta Mayor’s Office, 2021b).

One of the key strategies promoted by the Mayor’s Office in this regard has been the support of urban agriculture through grants, training, and increasing land access (Food Well Alliance, 2017; City of Atlanta Mayor’s Office, 2021a). By 2020, the population living in food deserts was reduced to 30% (City of Atlanta, 2020). Thus, it can be seen that urban farms, and their links to other community projects, may have facilitated this improved access (City of Atlanta, 2020). Many urban farms in Atlanta also provide other key social benefits in their local community; for example, working with healthcare services or providing educational and employment opportunities (Food Well Alliance, 2017). The social benefits of urban farms were also seen in the research undertaken with the Foodhall Project in Sheffield, UK (Chapter 5), where local farms were seen to be a key source of fresh fruits and vegetables for emergency food parcels during COVID-19. Thus, it becomes clear that, even though urban farms may not always be particularly resource efficient, they may also be providing important social benefits within cities.

Considering these layers of local context, it is vital for a given community to highlight and prioritise the aims of their local food system, because there will almost always be trade-offs. In Atlanta, one of the key aims was improving food access within the city limits, and urban farms were an important strategy toward this goal, even if they were not particularly ‘sustainable’ when compared to other peri-urban farms within the LCA. Other neighbourhoods or cities may find that different models of local agriculture better suit the population's needs. Thus, it is imperative to consider place-based contexts, within a wide food systems perspective, before promoting any food or agricultural strategy.

6.2.3 Supporting a multifunctional food system

For global food security to be achieved in the long-term, food systems must now support a range of environmental, social, and health-related goals (see: Introduction Section 1.1.1). Farmers, as a main foundation of the food system, are therefore being increasingly called upon to produce a range of services other than just food (Dimbleby, 2021). Farmers are framed as "stewards of the land" who have a "duty of care" to protect natural resources and manage land for the common good (Tovey, 2008). Of course, the increased pressure being placed upon them cannot come without additional support. We cannot ask farmers to provide multiple services, yet pay for only one, especially as climate change continues to threaten their livelihoods. Care must be taken to not over-burden this sector without facilitating the structural change that is necessary to operate a truly multifunctional food system. Supporting policy, research, and financial resources are required to ensure this transition happens in a just and sustainable way.

Agricultural policy is particularly important as this shapes the types of farms that persist and the resources available to a given land manager for change (Batie, 2009). One way to transition farmers toward the provisioning of multiple ecosystem services is to use public funding to pay farmers for providing public goods (Navarro and López-Bao, 2018; Scown, Brady and Nicholas, 2020). This is exactly what is now being implemented in the UK through the environmental land management schemes (ELMS) (Defra & Rural Payments Agency, 2021), which are replacing the basic payments of the EU's Common Agricultural Policy. ELMS is seen as a key step toward UK biodiversity and net-zero targets (UK Government, 2018; BEIS, 2021c; Defra & Natural England, 2022). The concept of using incentive payments to finance farmers on the basis of providing public goods (i.e., ecosystem services) can be viewed as quite a novel approach in comparison to other agricultural policies, such as in the EU and U.S., which are still slanted toward financial support based on land area, production, and/or efficiency rather than other environmental and social aims, even if these aims are part of overarching food policy (Batie, 2009; Navarro and López-Bao, 2018; Smith and Glauber, 2019).

In the U.S., the most important piece of legislation affecting the agricultural sector is what is commonly known as the Farm Bill (Batie, 2009); this is federally enacted every four to five years, most recently in 2018 (U.S. Congress, 2018). The majority of the Farm Bill's budget is allocated to subsidies for commodity crops, such as cotton, rice, corn, wheat, and soybeans, as well as dairy products and sugar (Smith and Glauber, 2019). Voluntary conservation programmes that aim to increase the protection of natural resources and meet environmental aims are also included within the Farm Bill, but these are poorly funded in comparison to commodity crop programmes (Batie, 2009; Stubbs, 2019). Some have thus argued that the Farm Bill has encouraged the expansion of agricultural land and the intensified production of commodity crops (Windham, 2007; Batie, 2009), whilst discouraging the production of 'healthier' crops like fruits and vegetables (Balagtas *et al.*, 2014).

In Europe, the main piece of legislature relevant to the agricultural sector is the EU's Common Agricultural Policy (CAP) (European Commission, 2023), which provides basic payments to farmers based mainly on land area or numbers of livestock (Ciaian and Swinnen, 2006; Navarro and López-Bao, 2018). This was also the presiding policy in the UK prior to exit from the EU, with basic payments now being phased out gradually by 2027 in favour of ELMS (Defra, 2022d; Defra & Rural Payments Agency, 2023). CAP influences land management on nearly half of the EU's terrestrial area, thus providing a key opportunity to promote environmental aims (Pe'er *et al.*, 2014). However, CAP has been criticised for not including ambitious enough environmental objectives (Pe'er *et al.*, 2014; Navarro and López-Bao, 2018; Dupraz and Guyomard, 2019), with the 'green payments' that did exist in the 2014-20 CAP seen to have low environmental effectiveness (European Court of Auditors, 2017). Like the U.S. Farm Bill, CAP has been linked to the intensification of agriculture, with landscape homogenisation and increased application of fertilisers and pesticides per hectare highlighted as contributing to widespread biodiversity loss in Europe (Emmerson *et al.*, 2016; Gamero *et al.*, 2017). This has led to calls for the CAP payments to be instead linked to environmental objectives, rather than production or farm area, similar to what is happening in the UK (Navarro and López-Bao, 2018).

The new ELMS policy in the UK can thus be seen as a potentially exciting way to use agricultural policy to meet environmental aims and to support farmers to provide ecosystem services for the common good. This forthcoming change in agricultural policy was seen to be particularly important in shaping farmers' interest to participate in the LCA study in the UK. This became apparent through the conversations had with all initially recruited farmers, who were asked about their motivations and interest in the study (see: Appendix D). While motivations for organic farmers across both countries were fairly similar, mainly wanting to test the sustainability of their farm models against more traditional modes of agriculture, what was particularly insightful was the different motivations between large-scale, conventional farmers in the U.S. and UK.

In Georgia, it was actually quite difficult to recruit farmers to the study, and those that did participate cited the main reason as wanting to help the researcher (who is also from the state). Contrastingly, recruitment was not an issue in the UK, with many conventional farmers showing interest in the study. This interest stemmed mainly from a desire to provide insight to their farm's carbon footprint, identify particular 'impact hot-spots,' and test the usefulness of certain 'environmentally-friendly' practices that they were implementing. Both during the LCA study as well as the soil health assessment, several conventional farmers highlighted the ways that they were attempting to improve sustainability on the farm, such as employing minimum tillage techniques, utilising solar energy, and using alternative packaging; one farm even eliminated the use of herbicides completely. Farmers cited upcoming ELMS policy, the National Farmers' Union target to achieve net-zero by 2040 (NFU, 2019), and increasing pressure from supermarket retailers for sustainability metrics as key drivers toward wanting to assess environmental impacts within the LCA study. In contrast, U.S. conventional farmers did not mention policy as a motivation. In addition, although U.S. supermarket chains have reported goals of supporting sustainable agriculture and increasing sustainability metric reporting throughout the supply chain (Kroger, 2022; Walmart, 2022), the conventional farmers in Georgia noted that they did not generally feel pressure from supermarkets retailers in this regard.

Although this evidence is largely anecdotal and considers only a very small portion of farms in both locations, these conversations highlight a key opportunity for agricultural policy to drive interest in evaluating and tracking environmental impacts on farms and employing sustainable agricultural practices. Clearly another opportunity is for consumers to act as an engine for change by increasing pressure on food retailers to better track and manage sustainability goals within their supply chains (Horton, Koh and Guang, 2016; Mangla *et al.*, 2018). Of course, this pressure needs to come with required support for farmers, else risking the over-burdening of an already financially constrained and mentally stressed sector (Naik, 2017). Indeed, a major concern of transitioning from CAP to ELMS is the financial loss that many farms may face, both from the overall lower public payments and potential increased costs to meet ELMS incentive requirements (Dimbleby, 2021; NFU, 2021c; AHDB, 2022a).

Finally, as we transition toward supporting farmers to provide a range of ecosystem services, it is important that small farmers are not left out. Both this study, as well as past studies, have highlighted that small-scale farms can provide a range of ecosystem and social services to their communities (Brenda B. Lin, Philpott and Jha, 2015; Laughton, 2017; Clinton *et al.*, 2018; Karlsson, Tidåker and Röös, 2022). However, agricultural policies have typically favoured larger-scale farms (Bekkerman *et al.*, 2019). Both the U.S. Farm Bill and CAP have been criticised as serving to enrich large landowners and spurring the concentration and corporatization of the agricultural sector (Ciaian and Swinnen, 2006; Troitino and Chochia, 2013; Graddy-Lovelace and Diamond, 2017; Bekkerman *et al.*, 2019). For example, as CAP basic payments are made on the basis of land area, this favours larger landowners; additionally, there is minimum threshold to receive these payments. In the UK, basic payments made to farmers under CAP apply only to farms with more than 5 ha of agricultural land (Defra & Rural Payments Agency, 2023). As seen within the work of this thesis, many urban and peri-urban farms in the UK were below this size. A minimum of 5 ha is also required to participate in initial Sustainable Farming Incentive pilots (as part of ELMS), although this is expected to be eliminated by 2024 (Defra & Rural Payments Agency, 2022b). In the U.S., there has been some national support for small farms, particularly seen with new grant programmes for urban agriculture following the COVID-19 pandemic (USDA, 2022; USDA Press, 2022).

Overall, it is clear that there must be policy change, as well as additional research and educational support for farms small and large to be able to meet the increasing demand for ecosystem services, reduced carbon emissions, and healthy, nutritious food.

6.2.4 The power in local

One of the main advantages of local food systems is the ability to provide this support and facilitate this change within local contexts, drawing on and uplifting the specific needs of producers and consumers within the community. The needs of these groups are often lost in the current global food system, where ownership and control are largely concentrated in a small number of transnational corporations, namely agrochemical and seed companies, food processors, and also retailers to some extent (Sexton, 2012; Graddy-Lovelace and Diamond, 2017; Clapp, 2021). One of the most drastic examples of this is for broiler chicken production in the U.S., where nearly all production is controlled by food processors through vertically-integrated supply chains; in some cases, farmers under contract with these firms do not own the live animal at all, thus severely limiting any ability to act independently (Goodwin, 2005; Sexton, 2012; Vukina and Zheng, 2015). This current era in food history has thus been referred to as the 'corporate food regime' (McMichael, 2005).

Considering the increasingly inequitable distribution of power within the food system, this has led calls for a complete upheaval of the corporate food system to put power back in the hands of the people (Sumner, 2011; Hendrickson, 2020). This is indeed a key aim of the food sovereignty movement, which emphasises that food is a basic human right and that people also have the right to control their own food systems and modes of production (Sage, 2014). Relocalising food systems is seen as a major part of this redistribution of power , as well as key to building resiliency (Sage, 2014; Robbins, 2015).

Herein lies some of the main benefits and opportunities for local food systems. As seen throughout this thesis, 'more local' food production is not always the most resource-efficient or productive. However, what producing food in and around cities provides is the opportunity for close connections to be made between producers, local retailers, consumers, community groups, and local government, so that these groups can work together to increase sustainability throughout the supply chain and address specific community needs (Campbell, 2004; Clayton *et al.*, 2015). The importance of this was highlighted in Chapter 5, where it became clear that the partnerships made between Foodhall and local shops, farms, other charities, and the city council were crucial to providing an emergency food supply for people across Sheffield – many of whom had been neglected by national support programmes. Facilitating this local food distribution effort meant that critical areas on the outskirts of Sheffield, which lacked other charity support (e.g., food banks), could be targeted to ensure that people were not left behind. Because Foodhall was already closely connected to local food actors and a suite of engaged volunteers prior to COVID, the organisation was able to swiftly transition from a café to an emergency food parcel delivery service during the first week of lockdown. However, had this not been the case, and had Sheffield not already had strong local food partnerships in place (ShefFood, 2023), this may not have been possible. Thus, this study and many others demonstrate the importance of both intra- and inter-connected local food system networks in building resiliency in the face of crisis (Lever, Sonnino and Cheetham, 2019; Hendrickson, 2020; Blay-Palmer *et al.*, 2021; Thilmany *et al.*, 2021; Jones, Krzywoszynska and Damian Maye, 2022; Turcu and Rotolo, 2022).

At the same time, the fact that Foodhall showed extreme adaptability to meet local needs during COVID, but was unable to sustain the financial loss that came with the pandemic, begs the question of how community spaces like this can be better supported in the future to maintain these crucial hubs of connection. We thus see a key opportunity for local authorities to become more involved in food policy to facilitate the transition toward more sustainable (and therefore just) local food systems. This allows for solutions to be developed that tackle the specific environmental, natural resource, and social challenges within a locality (Dankwa-Mullan and Pérez-Stable, 2016; Reece, 2018; Turcu and Rotolo, 2022). Land management decisions made on local scales can also better address ecosystem service needs and trade-offs, whilst coordinating within national frameworks and goals (Norton *et al.*, 2016). It is important that local food policy supports the development and continued provisioning of community food infrastructure which provides logistical and health support services to people, as well as the opportunities to build capabilities and social connections within a community, as this is crucial to building resilience (Béné, 2020). Finally, local food policy has a key opportunity to support farmers and landworkers as stewards of the land (Halliday, 2019), particularly those that may not typically benefit from national programmes otherwise (Sexton, 2012), such as small-scale or urban farms.

City governments have been key in developing local food policy in recent years to facilitate crisis response and target support toward specific place-based needs (Clark, Conley and Raja, 2021). This is seen through a variety of city food strategies, which support areas such as food security and access, food safety, social inclusion, food supply resilience, and food culture, among others (Calori and Magarini, 2015; Halliday, 2019). In many cases, these plans surpass national frameworks for responding to food-related challenges (Calori and Magarini, 2015). As previously referenced, Atlanta provides an optimal example of a city that has used local food policy to support urban farmers and address key food access and justice issues (City of Atlanta, 2020; City of Atlanta Mayor's Office, 2021a). In 2015, this city was the first in the U.S. to appoint an urban agriculture director in its Mayor's Office, with many other U.S. cities following suit soon after (Stephens, 2019). Since then, Atlanta has been home to a wide variety of innovative urban agriculture projects supported by the city government, such as the creation of a community food forest (Holcombe, 2019; AgLanta, 2022b) and the development of the AgLanta 'Grows-a-Lot' Program, which improves land access for urban farmers and community gardeners by providing opportunities to grow in power line easements for reduced land rent prices (AgLanta, 2022a).

Although relocalising food systems can provide a key opportunity to better serve the needs of a given community, it is important that this includes *all* the community. Indeed, a major critique of local farming models is that these tend to produce premium products that cater more to affluent consumers rather than serving ideals of food justice (Robbins, 2015); thus, encouraging more local agriculture will not necessarily result in improved food security or food access if this is not supported with other social strategies. It is certainly just as possible for power to be inequitably distributed within a local food system as it is in the global food system; thus, emphasis must be placed on building *inclusive* and *just* local food systems (Allen, 2010; Reece, 2018). At the same time, local food systems cannot and do not exist in complete isolation (Klassen and Wittman, 2017). Global trade is still required to ensure a consistent food supply, and any policies and actions taken to serve the needs of one community must also consider how this affects communities elsewhere.

6.2.5 Looking forward

The work of this thesis has focused on case studies in the U.S. and UK to provide insight to the sustainability and resiliency of local food systems. This was facilitated by the use of an inter-disciplinary, multi-method framework that allowed for the evaluation of environmental impacts, ecosystem services, and wellbeing factors across different spaces and different actors of the food system (Figure 74). Despite the integrated approach, this research still provides insight to only a small subset of food system challenges within a limited geographical scope. There are numerous other factors that influence the sustainability, resiliency, and applicability of local food systems that have not been explored.

For example, the evaluation of local food supply chains in this thesis focused mainly on different models of production, and how these comparatively contributed to food production, environmental impacts, and ecosystem services. Emphasis was placed on horticultural production, as this is the most common production type among urban and peri-urban farms and also a main focus for nutrition and health (De Bon, Parrot and Moustier, 2010; Boeing *et al.*, 2012; Kennard and Bamford, 2020). However, certain types of farms, such as those located in urban and peri-urban areas, may not be as suitable to produce other staple crops.

Thus, future research could also examine the viability of these models to produce different crop types and how different models may be combined to produce a wider range of foods with minimal impacts.

At the same time, an exaggerated focus on food production in and of itself can restrict the view of other equally vital and viable solutions to increase food supply and decrease environmental impacts, such as through changes in diet or reductions in food waste (Berners-Lee *et al.*, 2018; Kennard, 2019; Jarmul *et al.*, 2020). It has been found that employing these strategies, whilst also facilitating structural changes to increase food access, would negate the need to increase food production from current levels to meet global food demand in 2050 (Berners-Lee *et al.*, 2018). Thus, although agricultural production was the main focus of this thesis, it is indeed only one part of the puzzle (Tomlinson, 2013).

This thesis also focused mainly on environmental and social issues within the food system, albeit a limited set; in future work, this analysis would also include other socio-economic factors. For example, examining the economic viability of different local farm models would be of interest (Pölling *et al.*, 2017; Kafle, Hopeward and Myers, 2022), as well as considering how these contribute to local economies (e.g., providing quality jobs), potentially through the use of novel 'social' LCAs (De Luca *et al.*, 2015; Chen and Holden, 2017). Additionally, more detailed assessments are needed to examine if and how local food systems are shifting power dynamics to better support a common good, or if this is leading to any unintended consequences; for example, the contribution of urban agriculture to gentrification has been cited as a potential issue (McClintock, 2018; Reece, 2018).

Finally, examining new agri-environmental schemes that pay farmers for public goods, and how these collectively influence environmental outcomes, uptake of sustainable agricultural practices, and farmers' livelihoods, is of interest. This is particularly relevant in the UK with the recent implementation of ELMS, which has created concern about if the provided incentives will outweigh costs to farmers, or indeed just shift the burden (Dimbleby, 2021; NFU, 2021c). On a global scope, it will be crucial to consider how ecosystem services should be measured and valued. Indeed, the idea of placing a monetary value on ecosystem services has generated concerns about the commodification of nature's services, a marketising of the commons, and new exploitation through 'carbon colonialism' (Bachram, 2004; Sullivan, 2011; Böhm, Misoczky and Moog, 2012). This can particularly be seen through the burgeoning area of carbon offset trading; although some see this as a potentially important new income stream for farms (Tovey, 2008; AHDB, 2022b, 2022a; Green Alliance, 2022), others cite carbon markets as spurring a new land grab in the global South (Lyons and Westoby, 2014), or generally within rural communities across the globe, as companies buy and 'seal up' land to offset carbon emissions (White, 2013).

It is therefore important to note that, although this thesis focused mainly on opportunities and challenges for local food systems, this always sits within the inescapable context of the global food system (Klassen and Wittman, 2017; Lever, Sonnino and Cheetham, 2019). Food systems and ecosystems are inherently complex, with actions made in one part of the world reverberating through all others, sometimes with unintended consequences. If strategies are promoted to serve one population, or support one environmental aim, it is essential to consider and evaluate how this may impact others (Barbier and Heal, 2006). As food systems face an ever-growing array of complexities and challenges, it is crucial that researchers come together across disciplines to evaluate these challenges in more comprehensive and holistic ways.

6.3 Conclusion

Local food systems provide an important pathway toward increasing sustainability and resiliency. Although 'more local' food production does not always result in the lowest environmental impacts per crop output, the key advantages of increasing small-scale food production in and around cities are the ecosystem services and social advantages provided. (Re)localising food systems enable opportunities to shift control back to producers and consumers, address specific social and place-based needs, and build connections throughout the supply chain to build resiliency within the system. Building resiliency in the food system also means supporting a range of models to contribute to the wide array of services required from the food system, with their application tuned to local contexts and community needs whilst recognising global implications. Discourse needs to move beyond the divide in 'organic' vs. 'conventional' to recognise the positive contributions that all farms can make to their local communities, environments, and economies, thus creating policies and educational platforms that bring farmers together and collectively support the uptake of sustainable agricultural practices. Recognising that food production is only one component of food security, there also needs to be community food infrastructure in place that allows for people to come together during times of crisis and addresses the interconnected hardships of food insecurity, social isolation, and ill health. As we zoom out from the case studies of this thesis, we see that increased local control of food systems provides the opportunity to ensure justice throughout, so that access to food is truly a right and not just a privilege.

References

Abdulkadir, A. *et al.* (2013) ‘Nutrient flows and balances in urban and peri-urban agroecosystems of Kano, Nigeria’, *Nutrient Cycling in Agroecosystems*, 95(2), pp. 231–254. doi:10.1007/s10705-013-9560-2.

Ackerman, K. (2012) *The potential for urban agriculture in New York City: Growing capacity, food security, and green infrastructure*. New York City: Urban Design Lab, Columbia University.

Ackerman, K. *et al.* (2014) ‘Sustainable food systems for future cities: The potential of urban agriculture’, *The Economic and Social Review*, 45(2), pp. 189–206.

Adam, S., Miller, H. and Waters, T. (2020) *Income protection for the self-employed and employees during the coronavirus crisis*. London, UK: The Institute for Fiscal Studies. Available at: https://ifs.org.uk/sites/default/files/output_url_files/BN277-Income-protection-for-the-self-employed-and-employees-during-the-coronavirus-crisis.pdf.

Adams, M., Bell, L.A. and Griffin, P. (1997) *Teaching for diversity and social justice: A sourcebook*. New York, NY, USA: Routledge.

Aday, S. and Aday, M.S. (2020) ‘Impact of COVID-19 on the food supply chain’, *Food Quality and Safety*, 4(4), pp. 167–180. doi:10.1093/fqsafe/fyaa024.

ADEME (2012) *Programme de recherche de l’ADEME sur les émissions atmosphériques du compostage. Connaissances acquises et synthèse bibliographique*. Angers, France: French Environment and Energy Management Agency (ADEME). Available at: https://librairie.ademe.fr/cadic/3520/_84270__emissions_compostage.pdf.

ADEME (2020) *AGRIBALYSE v.3.0*. Angers, France. Available at: https://doc.agribalyse.fr/documentation-en/.

Ademiluyi, Y.S. *et al.* (2007) ‘Performance Assessment of Two- Wheel Tractor on a Loamy Sand Soil of Ilorin’, *Nigerian Journal of Technology*, 26(1), pp. 59–66.

Adetunji, A.T. *et al.* (2020) ‘Management impact and benefit of cover crops on soil quality: A review’, *Soil and Tillage Research*, 204(104717). doi:10.1016/j.still.2020.104717.

Adewale, C. *et al.* (2016) ‘Identifying hotspots in the carbon footprint of a small scale organic vegetable farm’, *Agricultural Systems*, 149, pp. 112–121. doi:10.1016/j.agsy.2016.09.004.

AgLanta (2022a) *AgLanta Grows-a-Lot Program*. Available at: https://www.aglanta.org/aglanta-grows-a-lot (Accessed: 27 January 2023).

AgLanta (2022b) *Urban Food Forest at Browns Mill*. Available at: https://www.aglanta.org/urban-food-forest-at-browns-mill-1 (Accessed: 27 January 2023).

AHDB (2015) *Opportunities for cover crops in conventional arable rotations. Information Sheet 41*. Agriculture and Horticulture Development Board. Available at: https://ahdb.org.uk/cover-crops.

AHDB (2022a) *Assessing the impact of the Sustainable Farming Incentive on farm businesses*. Kenilworth, UK: Agriculture and Horticulture Development Board. Available at: https://ahdb.org.uk/assessing-the-impact-of-the-sustainable-farming-incentive-on-farm-businesses.

AHDB (2022b) *Horizon blog: Are carbon markets a saving grace for farmers?, News - Agriculture and Horticulture Development Board*. Available at: https://ahdb.org.uk/trade-and-policy/horizon-blog-carbon-markets (Accessed: 26 January 2023).

Aldrich, D.P. and Meyer, M.A. (2015) ‘Social Capital and Community Resilience’, *American Behavioral Scientist*, 59(2), pp. 254–269. doi:10.1177/0002764214550299.

Alexander, P. *et al.* (2023) ‘High energy and fertilizer prices are more damaging than food export curtailment from Ukraine and Russia for food prices, health and the environment’, *Nature Food*, 4, pp. 84–95. doi:10.1038/s43016-022-00659-9.

Ali, M.Y. *et al.* (2021) ‘Nutritional composition and bioactive compounds in tomatoes and their impact on human health and disease: A review’, *Foods*, 10(1). doi:10.3390/foods10010045.

Alignier, A., Uroy, L. and Aviron, S. (2020) ‘The role of hedgerows in supporting biodiversity and other ecosystem services in intensively managed agricultural landscapes’, in Bàrberi, P. and Moonen, A.-C. (eds) *Reconciling agricultural production with biodiversity conservation*. 1st edn. London, UK: Burleigh Dodds Science Publishing, pp. 177–204. doi:10.19103/AS.2020.0071.09.

Allen, P. (2008) ‘Mining for justice in the food system: Perceptions, practices, and possibilities’, *Agriculture and Human Values*, 25(2), pp. 157–161. doi:10.1007/s10460-008-9120-6.

Allen, P. (2010) ‘Realizing justice in local food systems’, *Cambridge Journal of Regions, Economy and Society*, 3(2), pp. 295–308. doi:10.1093/cjres/rsq015.

Alloway, B.J. (2004) ‘Contamination of soils in domestic gardens and allotments: a brief overview’, *Land Contamination & Reclamation*, 12(3), pp. 179–187. doi:10.2462/09670513.658.

Alloway, B.J. (2013) ‘Sources of Heavy Metals and Metalloids in Soils’, in Alloway, B.J. (ed.) *Heavy metals in soils. Trace metals and metalloids in soils and their bioavailability*. 3rd edn. Dordrecht: Springer, pp. 11–50. doi:10.1007/978-94-007-4470-7.

Almeida, J. *et al.* (2014) ‘Carbon and water footprints and energy use of greenhouse tomato production in Northern Italy’, *Journal of Industrial Ecology*, 18(6), pp. 898–908. doi:10.1111/jiec.12169.

Almutairi, K. *et al.* (2015) ‘Life cycle assessment and economic analysis of residential air conditioning in Saudi Arabia’, *Energy and Buildings*, 102, pp. 370–379. doi:10.1016/j.enbuild.2015.06.004.

Altieri, M.A. (1995) *Agroecology: The Science of Sustainable Agriculture*. Boulder, CO: Westview Press.

Altieri, M.A. *et al.* (1999) ‘The greening of the “barrios”: Urban agriculture for food security in Cuba’, *Agriculture and Human Values*, 16, pp. 131–140. doi:10.1023/A.

Altieri, M.A. (2009) ‘Agroecology, small farms, and food sovereignty’, *Monthly Review*, 61(3), pp. 102–113.

Altieri, M.A. and Nicholls, C.I. (2018) ‘Urban Agroecology: designing biodiverse, productive and resilient city farms’, *Agro Sur*, 46(2), pp. 49–60. doi:10.4206/agrosur.2018.v46n2-07.

Altieri, M.A. and Toledo, V.M. (2011) ‘The agroecological revolution in Latin America: rescuing nature, ensuring food sovereignty and empowering peasants’, *Journal of Peasant Studies*, 38(3), pp. 587–612. doi:10.1080/03066150.2011.582947.

Amlinger, F., Peyr, S. and Cuhls, C. (2008) ‘Green house gas emissions from composting and mechanical biological treatment’, *Waste Management and Research*, 26(1), pp. 47–60. doi:10.1177/0734242X07088432.

Amlinger, F., Pollack, M. and Favoino, E. (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*. European Commission. Available at: https://ec.europa.eu/environment/pdf/waste/compost/hm_finalreport.pdf.

Amorim, H.C.S. *et al.* (2022) ‘C:N ratios of bulk soils and particle-size fractions: Global trends and major drivers’, *Geoderma*, 425(116026). doi:10.1016/j.geoderma.2022.116026.

Ander, E.L. *et al.* (2013) ‘Methodology for the determination of normal background concentrations of contaminants in English soil', *Science of The Total Environment*, 454–455, pp. 604–618. doi:10.1016/j.scitotenv.2013.03.005.

Andersen, J.K., Boldrin, A., Christensen, T.H., *et al.* (2010) 'Greenhouse gas emissions from home composting of organic household waste', *Waste Management*, 30(12), pp. 2475–2482. doi:10.1016/j.wasman.2010.07.004.

Andersen, J.K., Boldrin, A., Samuelsson, J., *et al.* (2010) 'Quantification of Greenhouse Gas Emissions from Windrow Composting of Garden Waste', *Journal of Environmental Quality*, 39(2), pp. 713–724. doi:10.2134/jeq2009.0329.

Andrews, S.S., Karlen, D.L. and Mitchell, J.P. (2002) 'A comparison of soil quality indexing methods for vegetable production systems in Northern California', *Agriculture, Ecosystems & Environment*, 90(1), pp. 25–45. doi:10.1016/S0167-8809(01)00174-8.

Antisari, L.V. *et al.* (2015) 'Heavy metal accumulation in vegetables grown in urban gardens.', *Agronomy for Sustainable Development*, 35(3), pp. 1139–1147. doi:10.1007/s13593-015-0308-z.

Antolini, F. *et al.* (2020) 'Flood Risk Reduction from Agricultural Best Management Practices', *JAWRA Journal of the American Water Resources Association*, 56(1), pp. 161–179. doi:10.1111/1752-1688.12812.

Antón, A. *et al.* (2004) 'Comparison of toxicological impacts of integrated and chemical pest management in Mediterranean greenhouses', *Chemosphere*, 54(8), pp. 1225–1235. doi:10.1016/j.chemosphere.2003.10.018.

Antón, A. *et al.* (2012) 'Environmental impact assessment of Dutch tomato crop production in a Venlo glasshouse', *Acta Horticulturae*, (927), pp. 781–791. doi:10.17660/ActaHortic.2012.927.97.

APESA, OLENTICA and BIO Intelligence Service (2015) *Impact sanitaire et environnementaux du compostage domestique – Rapport – Partie A*. Available at: www.ademe.fr,/mediatheque%0AToute.

Aranda, K. and Hart, A. (2014) 'Resilient moves: Tinkering with practice theory to generate new ways of thinking about using resilience', *Health: An Interdisciplinary Journal for the Social Study of Health, Illness and Medicine*, 19(4), pp. 355–371. doi:10.1177/1363459314554318.

Arcas-Pilz, V. *et al.* (2022) 'Extended use and optimization of struvite in hydroponic cultivation systems', *Resources, Conservation and Recycling*, 179. doi:10.1016/j.resconrec.2021.106130.

Ardente, F. and Cellura, M. (2012) 'Economic Allocation in Life Cycle Assessment', *Journal of Industrial Ecology*, 16(3), pp. 387–398. doi:10.1111/j.1530-9290.2011.00434.x.

Arrieta, T. (2022) 'Austerity in the United Kingdom and its legacy: Lessons from the COVID-19 pandemic', *Economic and Labour Relations Review*, 33(2), pp. 238–255. doi:10.1177/10353046221083051.

Arvela, P. (2013) 'Ethnic Food: The Other in Ourselves', in Sanderson, D. and Crouch, M. (eds) *Food: Expressions and Impressions*. Leiden, The Netherlands: Brill, pp. 43–56. doi:10.1163/9781848882140_006.

Arzoumanidis, I. *et al.* (2013) 'Life Cycle Assessment for the Agri-Food Sector', in Salomone, R. et al. (eds) *Product-Oriented Environmental Management Systems (POEMS)*. 1st edn. Dordrecht: Springer Netherlands, pp. 105–122. doi:10.1007/978-94-007-6116-2_5.

Atkinson, G. *et al.* (1997) *Measuring sustainable development: macroeconomics and the environment*. Cheltenham, UK: Edward Elgar.

Atlanta Regional Commission (2019) *Cities, Georgia*. Available at: https://opendata.atlantaregional.com/datasets/34520575dfc34b8cac783caff702b8cc_58?geometry=-103.827%2C30.535%2C-62.013%2C36.924 (Accessed: 20 December 2019).

Audsley, E. *et al.* (1997) *Harmonisation of Environmental Life Cycle Assessment for Agriculture*. Silsoe, UK: European Commission. Available at: https://cordis.europa.eu/project/id/AIR32028.

Audsley, E. *et al.* (2009) *How low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope to reduce them by 2050*. WWF-UK. Available at: https://dspace.lib.cranfield.ac.uk/handle/1826/6503.

Avadí, A. *et al.* (2020) 'Life cycle inventory data on French organic waste treatments yielding organic amendments and fertilisers', *Data in Brief*, 28. doi:10.1016/j.dib.2019.105000.

Avadí, A. *et al.* (2022) 'Suitability of operational N direct field emissions models to represent contrasting agricultural situations in agricultural LCA: Review and prospectus', *Science of the Total Environment*, 802(149960). doi:10.1016/j.scitotenv.2021.149960.

Aymard, V. and Botta-Genoulaz, V. (2017) 'Normalisation in life-cycle assessment: consequences of new European factors on decision-making', *Supply Chain Forum*, 18(2), pp. 76–83. doi:10.1080/16258312.2017.1333385.

Azevedo, L.B. *et al.* (2013) 'Global assessment of the effects of terrestrial acidification on plant species richness', *Environmental Pollution*, 174, pp. 10–15. doi:10.1016/j.envpol.2012.11.001.

Bachram, H. (2004) 'Climate fraud and carbon colonialism: The new trade in greenhouse gases', *Capitalism, Nature, Socialism*, 15(4), pp. 5–20. doi:10.1080/1045575042000287299.

De Backer, E. *et al.* (2009) 'Assessing the ecological soundness of organic and conventional agriculture by means of life cycle assessment (LCA) - a case study of leek production', *British Food Journal*, 111(10), pp. 1028–1061. doi:10.1108/00070700910992916.

Badgley, C. and Perfecto, I. (2007) 'Can organic agriculture feed the world?', *Renewable Agriculture and Food Systems*, 22(2), pp. 80–86. doi:10.1017/S1742170507001986.

Bai, L., Wang, M. and Gong, S. (2019) 'Understanding the Antecedents of Organic Food Purchases: The Important Roles of Beliefs, Subjective Norms, and Identity Expressiveness', *Sustainability*, 11(11), p. 3045. doi:10.3390/su11113045.

Bai, Z.G. *et al.* (2008) 'Proxy global assessment of land degradation', *Soil Use and Management*, 24, pp. 223–234. doi:10.1111/j.1475-2743.2008.00169.x.

Baker, E.A. *et al.* (2006) 'The role of race and poverty in access to foods that enable individuals to adhere to dietary guidelines', *Preventing Chronic Disease*, 3(3). Available at: http://www.cdc.gov/pcd/issues/ 2006/ jul/05_0217.htm.

Balagtas, J. V. *et al.* (2014) 'How has U.S. farm policy Influenced fruit and vegetable production?', *Applied Economic Perspectives and Policy*, 36(2), pp. 265–286. doi:10.1093/aepp/ppt028.

Balotin, L. *et al.* (2020) 'Atlanta residents' knowledge regarding heavy metal exposures and remediation in urban agriculture', *International Journal of Environmental Research and Public Health*, 17(2069). doi:10.3390/ijerph17062069.

Barbier, E.B. and Heal, G.M. (2006) 'Valuing Ecosystem Services', *The Economists' Voice*, 3(3), pp. 314–326. doi:10.2202/1553-3832.1118.

Barbosa, G.L. *et al.* (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, pp. 6879–6891. doi:10.3390/ijerph120606879.

Barker, M. and Russell, J. (2020) 'Feeding the food insecure in Britain: learning from the 2020 COVID-19 crisis', *Food Security*, pp. 10–12. doi:10.1007/s12571-020-01080-5.

Barrett, C.B. (2010) 'Measuring food insecurity', *Science*, 327(5967), pp. 825–828. doi:10.1126/science.1182768.

Bartfeld, J. and Collins, J.M. (2017) ‘Food Insecurity, Financial Shocks, and Financial Coping Strategies among Households with Elementary School Children in Wisconsin’, *Journal of Consumer Affairs*, 51(3), pp. 519–548. doi:10.1111/joca.12162.

Bassett, T.J. (1981) ‘Reaping on the Margins: A Century of Community Gardening in America’, *Landscape*, 25(2), pp. 1–8.

Batie, S.S. (2008) ‘Wicked problems and applied economics’, *American Journal of Agricultural Economics*, 90(5), pp. 1176–1191. doi:10.1111/j.1467-8276.2008.01202.x.

Batie, S.S. (2009) ‘Green payments and the US Farm Bill: information and policy challenges’, *Frontiers in Ecology and the Environment*, 7(7), pp. 380–388. doi:10.1890/080004.

Batson, C.D. and Shaw, L.L. (1991) ‘Evidence for Altruism: Toward a Pluralism of Prosocial Motives’, *Psychological Inquiry*, 2(2), pp. 107–122. doi:10.1207/s15327965pli0202_1.

Battersby, J. (2011) ‘Urban food insecurity in Cape Town, South Africa: An alternative approach to food access’, *Development Southern Africa*, 28(4), pp. 545–561. doi:10.1080/0376835X.2011.605572.

Bauer, M., Barney, D.L. and Robbins, J.A. (2009) *Growing tomatoes in cool, short-season locations: Bulletin 864, Short-Season, High-Altitude Gardening*. Moscow, Idaho. Available at: https://www.plantgrower.org/uploads/6/5/5/4/65545169/growing_tomatoes_in_cool_-_short_season_locations.pdf.

Baumann, H. and Tillman, A. (2004) *The Hitch Hiker’s guide to LCA: an orientation in life cycle assessment methodology and application*. Lund, Sweden: Studentlitteratur AB.

Bayer (2019) *Temperature, Humidity, and Water in Protected Culture Tomatoes: Cultivation Insights*. Available at: https://www.vegetables.bayer.com/us/en-us/resources/growing-tips-and-innovation-articles/cultivation-insights/temperature-humidity-and-water-in-protected-culture-tomatoes.html.

Beare, M.H. *et al.* (1995) ‘A hierarchical approach to evaluating the significance of soil biodiversity to biogeochemical cycling’, *Plant and Soil*, 170(1), pp. 5–22. Available at: https://www.jstor.org/stable/42947356.

Beck, D.J. and Gwilym, H. (2022) ‘The Food Bank: A Safety-Net in Place of Welfare Security in Times of Austerity and the Covid-19 Crisis’, *Social Policy and Society*, pp. 1–17. doi:10.1017/S1474746421000907.

Becker, L.C. (2005) ‘Reciprocity, Justice, and Disability’, *Ethics*, 116(1), pp. 9–39. doi:10.1086/453150.

Beddington, J. (2010) *Food, energy, water and the climate: a perfect storm of global events?* London, UK: Government Office for Science. Available at: https://www.bl.uk/collection-items/food-energy-water-and-the-climate-a-perfect-storm-of-global-events.

Begley, A. *et al.* (2019) ‘Examining the Association between Food Literacy and Food Insecurity’, *Nutrients*, 11(2), p. 445. doi:10.3390/nu11020445.

BEIS (2020) *UK Energy in Brief 2020*. London, UK: UK Department for Business, Energy & Industrial Strategy. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/904503/UK_Energy_in_Brief_2020.pdf.

BEIS (2021a) *Combined Heat and Power – Environmental. A detailed guide for CHP developers – Part 3*. London, UK: UK Department for Business, Energy & Industrial Strategy. Available at: https://www.gov.uk/government/collections/combined-heat-and-power-chp-developers-guides#chp-development-guides.

BEIS (2021b) *Combined Heat and Power – Technologies. A detailed guide for CHP developers –*

Part 2. London, UK: UK Department for Business, Energy & Industrial Strategy. Available at: https://www.gov.uk/government/collections/combined-heat-and-power-chp-developers-guides#chp-development-guides.

BEIS (2021c) *Net Zero Strategy: Build Back Greener*. London, UK: UK Department for Business, Energy & Industrial Strategy. Available at: https://www.gov.uk/government/publications/net-zero-strategy.

Bekkerman, A. *et al.* (2019) 'Does Farm Size Matter? Distribution of Crop Insurance Subsidies and Government Program Payments across U.S. Farms', *Applied Economic Perspectives and Policy*, 41(3), pp. 498–518. doi:10.1093/aepp/ppy024.

Bellamy, A.S. *et al.* (2021) 'Shaping more resilient and just food systems: Lessons from the COVID-19 Pandemic', *Ambio*, 50(4), pp. 782–793. doi:10.1007/s13280-021-01532-y.

Béné, C. (2020) 'Resilience of local food systems and links to food security – A review of some important concepts in the context of COVID-19 and other shocks', *Food Security*, 12(4), pp. 805–822. doi:10.1007/s12571-020-01076-1.

Bengtsson, J., Ahnstrom, J. and Weibull, A.-C. (2005) 'The effects of organic agriculture on biodiversity and abundance: a meta-analysis', *Journal of Applied Ecology*, 42(2), pp. 261–269. doi:10.1111/j.1365-2664.2005.01005.x.

Benis, K. and Ferrão, P. (2017) 'Potential mitigation of the environmental impacts of food systems through urban and peri-urban agriculture (UPA) – a life cycle assessment approach', *Journal of Cleaner Production*, 140, pp. 784–795. doi:10.1016/j.jclepro.2016.05.176.

Beniston, J. and Lal, R. (2012) 'Improving Soil Quality for Urban Agriculture in the North Central U.S.', in Lal, R. and Augustin, B. (eds) *Carbon Sequestration in Urban Ecosystems*. Dordrecht: Springer Netherlands, pp. 279–313. doi:10.1007/978-94-007-2366-5.

Beniston, J., Lal, R. and Mercer, K. (2016) 'Assessing and Managing Soil Quality for Urban Agriculture', *Land Degradation & Development*, 27, pp. 996–1006. doi:10.1002/ldr.2342.

Bennett, L.T. *et al.* (2010) 'Examining links between soil management, soil health, and public benefits in agricultural landscapes: An Australian perspective', *Agriculture, Ecosystems and Environment*, 139, pp. 1–12. doi:10.1016/j.agee.2010.06.017.

Benton, T.G. (2020) 'COVID-19 and disruptions to food systems', *Agriculture and Human Values*, 37(3), pp. 577–578. doi:10.1007/s10460-020-10081-1.

Benton, T.G., Vickery, J.A. and Wilson, J.D. (2003) 'Farmland biodiversity: is habitat heterogeneity the key?', *Trends in Ecology & Evolution*, 18(4), pp. 182–188. doi:10.1016/S0169-5347(03)00011-9.

Berlant, L. (2007) 'Slow Death (Sovereignty, Obesity, Lateral Agency)', *Critical Inquiry*, 33(4), pp. 754–780. doi:10.1086/521568.

Berlant, L. (2011) *Cruel Optimism*. London, UK: Duke University Press. doi:10.1215/9780822394716.

Berlin, D. and Uhlin, H.-E. (2004) 'Opportunity cost principles for life cycle assessment: toward strategic decision making in agriculture', *Progress in Industrial Ecology, An International Journal*, 1(1–3), pp. 187–202. doi:10.1504/PIE.2004.004678.

Bernard, B. and Lux, A. (2017) 'How to feed the world sustainably: an overview of the discourse on agroecology and sustainable intensification', *Regional Environmental Change*, 17, pp. 1279–1290. doi:10.1007/s10113-016-1027-y.

Berners-Lee, M. *et al.* (2018) 'Current global food production is sufficient to meet human nutritional needs in 2050 provided there is radical societal adaptation', *Elementa Science of the Anthropocene*, 6(52). doi:https://doi.org/10.1525/elementa.310 RESEARCH.

Berrow, M.L. and Webber, J. (1972) ‘Trace elements in sewage sludges’, *Journal of the Science of Food and Agriculture*, 23(1), pp. 93–100. doi:10.1002/jsfa.2740230112.

Berry, P.M. *et al.* (2002) ‘Is the productivity of organic farms restricted by the supply of available nitrogen?’, *Soil Use and Management*, 18, pp. 248–255. doi:10.1111/j.1475-2743.2002.tb00266.x.

Beus, C.E. and Dunlap, R.E. (1990) ‘Conventional versus Alternative Agriculture: The Paradigmatic Roots of the Debate’, *Rural Sociology*, 55(4), pp. 590–616. doi:10.1111/j.1549-0831.1990.tb00699.x.

Beutel, M.E. *et al.* (2017) ‘Loneliness in the general population: Prevalence, determinants and relations to mental health’, *BMC Psychiatry*, 17(1), pp. 1–7. doi:10.1186/s12888-017-1262-x.

Bezner Kerr, R., Hasegawa, T. *et al.* (2022) ‘Food, Fibre, and Other Ecosystem Products’, in Pörtner, H.O. et al. (eds) *Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK and New York, NY, USA: Cambridge University Press, pp. 713–906. doi:10.1017/9781009325844.007.

Bhatia, M. (2020) ‘COVID-19 and BAME Group in the United Kingdom’, *The International Journal of Community and Social Development*, 2(2), pp. 269–272. doi:10.1177/2516602620937878.

Bian, Y., Hao, M. and Li, Y. (2018) ‘Social Networks and Subjective Well-Being: A Comparison of Australia, Britain, and China’, *Journal of Happiness Studies*, 19(8), pp. 2489–2508. doi:10.1007/s10902-017-9926-2.

Biodynamic Association (2019) *BDA Certification*. Available at: https://bdcertification.org.uk/index.php/bd-certification/ (Accessed: 20 December 2019).

Biroli, P. *et al.* (2021) ‘Family Life in Lockdown’, *Frontiers in Psychology*, 12. doi:10.3389/fpsyg.2021.687570.

Bjarnadóttir, H.J. *et al.* (2002) *Guidelines for the use of LCA in the waste management sector*. Fenur, Iceland: Nordtest. Available at: http://www.nordtest.org/register/techn/tlibrary/tec517/tec517.pdf.

Bjørn, A. *et al.* (2017) ‘Main Characteristics of LCA’, in Hauschild, M.Z., Rosenbaum, R.K., and Olsen, S.I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham, Switzerland: Springer International Publishing AG. doi:10.1007/978-3-319-56475-3_6.

Björnwall, A. *et al.* (2021) ‘Eating Alone or Together among Community-Living Older People—A Scoping Review’, *International Journal of Environmental Research and Public Health*, 18(7), p. 3495. doi:10.3390/ijerph18073495.

Black, C.A. (1993) *Soil fertility evaluation and control*. Boca Raton, FL, USA: Lewis Publishers.

Blake, M. (2018) ‘Landscape and the politics of food justice’, in Zeunert, J. and Waterman, T. (eds) *Routledge Handbook of Landscape and Food*. 1st edn. London, UK: Routledge, pp. 487–499. doi:10.4324/9781315647692-33.

Blake, M. (2019) ‘More than Just Food: Food Insecurity and Resilient Place Making through Community Self-Organising’, *Sustainability*, 11(10), p. 2942. doi:10.3390/su11102942.

Blake, M. (2021) ‘Building post-COVID community resilience by moving beyond emergency food support’, in Bryson, J.R. et al. (eds) *Living with Pandemics: Places, People and Policy*. Edward Elgar Publishing, pp. 59–68.

Blanco-Canqui, H. *et al.* (2015) ‘Cover crops and ecosystem services: Insights from studies in temperate soils’, *Agronomy Journal*, 107(6), pp. 2449–2474. doi:10.2134/agronj15.0086.

Blanco-Canqui, H., Francis, C.A. and Galusha, T.D. (2017) ‘Does organic farming accumulate carbon in deeper soil profiles in the long term?’, *Geoderma*, 288, pp. 213–221. doi:10.1016/j.geoderma.2016.10.031.

Blanco-Canqui, H. and Ruis, S.J. (2020) ‘Cover crop impacts on soil physical properties: A review’, *Soil Science Society of America Journal*, 84(5), pp. 1527–1576. doi:10.1002/saj2.20129.

Blanke, M.M. and Burdick, B. (2005) ‘Food (miles) for Thought: Energy balance for locally-grown versus imported apple fruit’, *Environmental Science and Pollution Research*, 12(3), pp. 125–127. doi:10.1065/espr2005.05.252.

Di Blasi, C., Tanzi, V. and Lanzetta, M. (1997) ‘A study on the production of agricultural residues in Italy’, *Biomass and Bioenergy*, 12(5), pp. 321–331. doi:10.1016/S0961-9534(96)00073-6.

Blay-Palmer, A. *et al.* (2021) ‘City region food systems: Building resilience to COVID-19 and other shocks’, *Sustainability (Switzerland)*, 13(3), pp. 1–19. doi:10.3390/su13031325.

Blengini, G.A. (2008) ‘Using LCA to evaluate impacts and resources conservation potential of composting: A case study of the Asti District in Italy’, *Resources, Conservation and Recycling*, 52(12), pp. 1373–1381. doi:10.1016/j.resconrec.2008.08.002.

Blonk, H. *et al.* (2010) *Methodology for assessing carbon footprints of horticultural products*. Gouda, Netherlands: Blonk Milieu Advies BV.

Blonk Sustainability (2017) *Agri-footprint v.4.0*. Gouda, The Netherlands. Available at: https://blonksustainability.nl/tools/agri-footprint.

Blundell, J. and Machin, S. (2020) *Self-employment in the Covid-19 crisis*. London, UK. Available at: https://cep.lse.ac.uk/pubs/download/cepcovid-19-003.pdf.

Bockstaller, C. and Girardin, P. (2010) *Mode de calcul des indicateurs Agri-environnementaux de la methode Indigo®*. Colmar, France: INRA. Available at: http://wiki.inra.fr/wiki/deximasc/download/package+MASC/WebHome/ManuelCalculIndicateurINDIGO.pdf.

Boeing, H. *et al.* (2012) ‘Critical review: vegetables and fruit in the prevention of chronic diseases’, *European Journal of Nutrition*, 51(6), pp. 637–663. doi:10.1007/s00394-012-0380-y.

Bofill, S. (2004) ‘Aging and loneliness in Catalonia: The social dimension of food behavior’, *Ageing International*, 29(4), pp. 385–398. doi:10.1007/s12126-004-1006-3.

Bogart, K.R. and Dunn, D.S. (2019) ‘Ableism Special Issue Introduction’, *Journal of Social Issues*, 75(3), pp. 650–664. doi:10.1111/josi.12354.

Böhm, S., Misoczky, M.C. and Moog, S. (2012) ‘Greening Capitalism? A Marxist Critique of Carbon Markets’, *Organization Studies*, 33(11), pp. 1617–1638. doi:10.1177/0170840612463326.

Bojacá, C.R., Wyckhuys, K.A.G. and Schrevens, E. (2014) ‘Life cycle assessment of Colombian greenhouse tomato production based on farmer-level survey data’, *Journal of Cleaner Production*, 69(3), pp. 26–33. doi:10.1016/j.jclepro.2014.01.078.

Bolan, N.S. and Duraisamy, V.P. (2003) ‘Role of inorganic and organic soil amendments on immobilisation and phytoavailability of heavy metals: a review involving specific case studies’, *Soil Research*, 41(3), p. 533. doi:10.1071/SR02122.

Boldrin, A. *et al.* (2009) ‘Composting and compost utilization: Accounting of greenhouse gases and global warming contributions’, *Waste Management and Research*, 27(8), pp. 800–812. doi:10.1177/0734242X09345275.

De Bon, H., Parrot, L. and Moustier, P. (2010) ‘Sustainable urban agriculture in developing countries. A review’, *Agronomy for Sustainable Development*, 30, pp. 21–32. doi:10.1017/S1751731110000674.

Bonanomi, G. *et al.* (2017) ‘Biochar As Plant Growth Promoter: Better Off Alone or Mixed with Organic Amendments?’, *Frontiers in Plant Science*, 8. doi:10.3389/fpls.2017.01570.

Boneta, A. *et al*. (2019) ‘Agronomic and Environmental Assessment of a Polyculture Rooftop Soilless Urban Home Garden in a Mediterranean City’, *Frontiers in Plant Science*, 10(341). doi:10.3389/fpls.2019.00341.

Bonevac, D. (2010) ‘Is Sustainability Sustainable?’, *Academic Questions*, 23(1), pp. 84–101. doi:10.1007/s12129-009-9152-4.

Boone, L. *et al*. (2019) ‘Environmental sustainability of conventional and organic farming: Accounting for ecosystem services in life cycle assessment’, *Science of the Total Environment*, 695(133841). doi:10.1016/j.scitotenv.2019.133841.

Bos, A.E.R. *et al*. (2013) ‘Stigma: Advances in Theory and Research’, *Basic and Applied Social Psychology*, 35(1). doi:10.1080/01973533.2012.746147.

Bos, J.F.F.P. *et al*. (2014) ‘Energy use and greenhouse gas emissions in organic and conventional farming systems in the Netherlands’, *NJAS - Wageningen Journal of Life Sciences*, 68, pp. 61–70. doi:10.1016/j.njas.2013.12.003.

Bot, A. and Benites, J. (2005) *The importance of soil organic matter: key to drought-resistant soil and sustained food production*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://www.fao.org/3/a0100e/a0100e00.htm.

Boulard, T. *et al*. (2011) ‘Environmental impact of greenhouse tomato production in France’, *Agronomy for Sustainable Development*, 31(4), pp. 757–777. doi:10.1007/s13593-011-0031-3.

Bourdieu, P. (1993) *Sociology in Question*. London, UK: Sage Publications Ltd.

Boustani, A. *et al*. (2010) *Appliance Remanufacturing and Energy Savings*. Cambridge, MA: Massachusetts Institute of Technology. Available at: https://web.mit.edu/ebm/www/Publications/MITEI-1-a-2010.pdf.

Bowles, D.C., Butler, C.D. and Morisetti, N. (2015) ‘Climate change, conflict and health’, *Journal of the Royal Society of Medicine*, 108(10), pp. 390–395. doi:10.1177/0141076815603234.

Boyle, N.B. and Power, M. (2021) ‘Proxy longitudinal indicators of household food insecurity in the UK’, *Emerald Open Research*, 3(16). doi:10.35241/emeraldopenres.14311.1.

Bradley, R.I. *et al*. (2005) ‘A soil carbon and land use database for the United Kingdom’, *Soil Use and Management*, 21(4), pp. 363–369. doi:10.1079/SUM2005351.

Bramley, G. *et al*. (2021) *State of Hunger: Building the evidence on poverty, destitution, and food insecurity in the UK. Year two main report*. The Trussell Trust. Available at: https://www.trusselltrust.org/wp-content/uploads/sites/2/2021/05/State-of-Hunger-2021-Report-Final.pdf.

Bridges, E.M. and Oldeman, L.R. (1999) ‘Global assessment of human-induced soil degradation’, *Arid Soil Research and Rehabilitation*, 13(4), pp. 319–325. doi:10.1080/089030699263212.

Briggs, S. and Foord, M. (2017) ‘Food Banks and the Transformation of British Social Welfare’, *ERIS Journal*, 17(4), pp. 72–86.

British Geological Survey (2001) *Soil Parent Material Model*, *BGS Datasets*. Available at: https://www.bgs.ac.uk/datasets/soil-parent-material-model/ (Accessed: 5 January 2023).

British Geological Survey (2021) *Soil property data*. Available at: https://www.bgs.ac.uk/technologies/web-map-services-wms/soil-property-data-wms/ (Accessed: 18 August 2022).

Brockmann, D., Pradel, M. and Hélias, A. (2018) ‘Agricultural use of organic residues in life cycle assessment: Current practices and proposal for the computation of field emissions and of the nitrogen mineral fertilizer equivalent’, *Resources, Conservation and Recycling*, 133, pp. 50–62.

doi:10.1016/j.resconrec.2018.01.034.

Brown, B.J. and Baker, S. (2013) *Responsible citizens: Individuals, health, and policy under neoliberalism*. Anthem Press.

Brown, H. and Reid, K. (2021) 'Navigating infodemics, unlocking social capital and maintaining food security during the COVID-19 first wave in the UK: Older adults' experiences', *International Journal of Environmental Research and Public Health*, 18(14). doi:10.3390/ijerph18147220.

Brown, J. and Kirk-Wade, E. (2021) *Coronavirus: A hisotry of 'Lockdown laws' in England*. London, UK: House of Commons Library. Available at: https://commonslibrary.parliament.uk/research-briefings/cbp-9068/.

Brown, K.H. and Jameton, A.L. (2000) 'Public Health Implications of Urban Agriculture', *Journal of Public Health Policy*, 21(1), pp. 20–39. doi:10.2307/3343472.

Brundtland, G.H. (1987) *Our Common Future: Report of the World Commission on Environment and Development*. Oxford, UK: Oxford University Press.

Bruun, S. *et al.* (2006) 'Application of processed organic municipal solid waste on agricultural land - A scenario analysis', *Environmental Modeling and Assessment*, 11(3), pp. 251–265. doi:10.1007/s10666-005-9028-0.

BSI (2011) *PAS 2050:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services.* London: British Standards Institution.

BSI (2012) *PAS 2050-1:2012 - Assessment of life cycle greenhouse gas emissions from horticultural products*. British Standards Institution.

Bu, F., Mak, H.W. and Fancourt, D. (2021) 'Rates and predictors of uptake of mental health support during the COVID-19 pandemic: an analysis of 26,720 adults in the UK in lockdown', *Social Psychiatry and Psychiatric Epidemiology*, 56(12), pp. 2287–2297. doi:10.1007/s00127-021-02105-w.

Bullen, E.R. (1974) *Burning Cereal Crop Residues in England*. London, UK: UK Ministry of Agriculture Fisheries and Food. Available at: https://agris.fao.org/agris-search/search.do?recordID=US201303193712.

Bullock, D.G. (1992) 'Crop rotation', *Critical Reviews in Plant Sciences*, 11(4), pp. 309–326. doi:10.1080/07352689209382349.

Burn, W. and Mudholkar, S. (2020) 'Impact of COVID-19 on mental health: Update from the United Kingdom', *Indian Journal of Psychiatry*, 62(Suppl 3), pp. S365–S372. doi:10.4103/psychiatry.IndianJPsychiatry_937_20.

Burns, C. *et al.* (2011) 'Reduced food access due to a lack of money, inability to lift and lack of access to a car for food shopping: a multilevel study in Melbourne, Victoria', *Public Health Nutrition*, 14(6), pp. 1017–1023. doi:10.1017/S136898001000385X.

Burns, R. (2014) 'Atlanta's food deserts leave its poorest citizens stranded and struggling', *The Guardian*, 17 March. Available at: https://www.theguardian.com/cities/2014/mar/17/atlanta-food-deserts-stranded-struggling-survive.

Busby, M. (2020) 'How coronavirus has led to a UK boom in community food growing', *The Guardian*, 24 August. Available at: https://www.theguardian.com/world/2020/aug/24/how-coronavirus-has-led-to-a-uk-boom-in-community-food-growing.

Butler, M.J. and Barrientos, R.M. (2020) 'The impact of nutrition on COVID-19 susceptibility and long-term consequences', *Brain, Behavior, and Immunity*, 87, pp. 53–54. doi:10.1016/j.bbi.2020.04.040.

Cacioppo, J.T., Hawkley, L.C. and Thisted, R.A. (2010) 'Perceived social isolation makes me sad: 5-

year cross-lagged analyses of loneliness and depressive symptomatology in the Chicago Health, Aging, and Social Relations Study.', *Psychology and Aging*, 25(2), pp. 453–463. doi:10.1037/a0017216.

Cahill, S., Morley, K. and Powell, D.A. (2010) 'Coverage of organic agriculture in North American newspapers', *British Food Journal*, 112(7), pp. 710–722. doi:10.1108/00070701011058244.

California Air Resources Board (2017) *Method for estimating greenhouse gas emission reductions from diversion of organic waste from landfills to compost facilities*. Available at: https://ww2.arb.ca.gov/sites/default/files/classic/cc/waste/cerffinal.pdf.

Calori, A. and Magarini, A. (2015) *Food and the cities: Food policies for sustainable cities*. Milan, Italy: Edizioni Ambiente.

Campbell, F. (2001) 'Inciting legal fictions: "Disability's" date with ontology and the ableist body of the law', *Griffith Law Review*, 10(1), pp. 42–62.

Campbell, M.C. (2004) 'Building a common table: The role for planning in community food systems', *Journal of Planning Education and Research*, 23(4), pp. 341–355. doi:10.1177/0739456X04264916.

Caplan, P. (2016) 'Big society or broken society? Food banks in the UK', *Anthropology Today*, 32(1), pp. 5–9. doi:10.1111/1467-8322.12223.

Caplan, P. (2020) 'Struggling for food in a time of crisis: Responsibility and paradox', *Anthropology Today*, 36(3), pp. 8–10. doi:10.1111/1467-8322.12573.

Cardoso, E.J.B.N. *et al.* (2013) 'Soil health: looking for suitable indicators. What should be considered to assess the effects of use and management on soil health ?', *Scientia Agricola*, 70(4), pp. 274–289.

Carpenter, S.R. *et al.* (2009) 'Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment', *Proceedings of the National Academy of Sciences*, 106(5), pp. 1305–1312. doi:10.1073/pnas.0808772106.

Carrano, A.L., Thorn, B.K. and Woltag, H. (2014) 'Characterizing the carbon footprint of wood pallet logistics', *Forest Products Journal*, 64(7–8), pp. 232–241. doi:10.13073/FPJ-D-14-00011.

Cattell, V. *et al.* (2008) 'Mingling, observing, and lingering: Everyday public spaces and their implications for well-being and social relations', *Health and Place*, 14(3), pp. 544–561. doi:10.1016/j.healthplace.2007.10.007.

CDH Energy Corp (2009) *Evaluation of the Coolbot Low-cost Walk-in Cooler Concept*. Albany, NY: New York State Energy Research and Development Authority. Available at: https://storeitcold.com/wp-content/uploads/2020/03/NYSERDA-CoolBot-Report-May-09.pdf.

Certified Naturally Grown (2023) *Certification types*. Available at: https://naturallygrown.org/certification-types/ (Accessed: 30 January 2023).

Chan, K.Y., Stoové, M.A. and Reidpath, D.D. (2008) 'Stigma, social reciprocity and exclusion of HIV/AIDS patients with illicit drug histories: A study of Thai nurses' attitudes', *Harm Reduction Journal*, 5(28). doi:10.1186/1477-7517-5-28.

Chander, K. and Brookes, P.C. (1991) 'Effects of heavy metals from past applications of sewage sludge on microbial biomass and organic matter accumulation in a sandy loam and silty loam U.K. soil', *Soil Biology and Biochemistry*, 23(10), pp. 927–932. doi:10.1016/0038-0717(91)90172-G.

Chaney, R.L., Sterret, S.B. and Mielke, H.W. (1984) 'The potentail for heavy metal exposure from urban gardens and soils', in Preer, J.R. (ed.) *Proc. Symp. Heavy Metals in Urban Gardens*. Washington, DC: Univ. Dist. Columbia Extension Service, pp. 37–84.

Chase, E. and Walker, R. (2013) ‘The Co-construction of Shame in the Context of Poverty: Beyond a Threat to the Social Bond’, *Sociology*, 47(4), pp. 739–754. doi:10.1177/0038038512453796.

Chen, B. *et al.* (2014) ‘Soil nitrogen dynamics and crop residues. A review’, *Agronomy for Sustainable Development*, 34(2), pp. 429–442. doi:10.1007/s13593-014-0207-8.

Chen, Jie *et al.* (2002) ‘Soil degradation: a global problem endangering sustainable development’, *Journal of Geographical Sciences*, 12(2), pp. 243–252. doi:10.1007/BF02837480.

Chen, W. and Holden, N.M. (2017) ‘Social life cycle assessment of average Irish dairy farm’, *The International Journal of Life Cycle Assessment*, 22(9), pp. 1459–1472. doi:10.1007/s11367-016-1250-2.

Cherfi, A., Abdoun, S. and Gaci, O. (2014) ‘Food survey: Levels and potential health risks of chromium, lead, zinc and copper content in fruits and vegetables consumed in Algeria’, *Food and Chemical Toxicology*, 70, pp. 48–53. doi:10.1016/j.fct.2014.04.044.

Chiffoleau, Y. and Dourian, T. (2020) ‘Sustainable Food Supply Chains: Is Shortening the Answer? A Literature Review for a Research and Innovation Agenda’, *Sustainability*, 12(23), p. 9831. doi:10.3390/su12239831.

Childers, T.B. (2005) *The effect of low and high fertility treatments on soil quality, yields, pest incidence and labor requirements of a post-transitional organic market garden system*. West Virginia University Libraries. doi:10.33915/etd.2285.

Chouinard, V. (1997) ‘Making space for disabling differences: Challenging ableist geographies’, *Environment and Planning D*, 15, pp. 379–387.

Christou, A. *et al.* (2014) ‘Impact assessment of the reuse of two discrete treated wastewaters for the irrigation of tomato crop on the soil geochemical properties, fruit safety and crop productivity’, *Agriculture, Ecosystems and Environment*, 192, pp. 105–114. doi:10.1016/j.agee.2014.04.007.

Chung, Y.C. (2007) ‘Evaluation of gas removal and bacterial community diversity in a biofilter developed to treat composting exhaust gases’, *Journal of Hazardous Materials*, 144(1–2), pp. 377–385. doi:10.1016/j.jhazmat.2006.10.045.

Ciaian, P. and Swinnen, J.F.M. (2006) ‘Land Market Imperfections and Agricultural Policy Impacts in the New EU Member States: A Partial Equilibrium Analysis’, *American Journal of Agricultural Economics*, 88(4), pp. 799–815. doi:10.1111/j.1467-8276.2006.00899.x.

City of Atlanta (2020) *Fresh Food Access Report 2020*. Atlanta, GA, USA. Available at: https://www.aglanta.org/2020-fresh-food-access-report.

City of Atlanta (2022) *AgLanta Grown*. Available at: https://www.aglanta.org/aglanta-grown (Accessed: 21 December 2022).

City of Atlanta Mayor’s Office (2021a) *City of Atlanta’s New Policy to Allow Urban Farms to Sell Directly to Consumers Adoption of farm stand ordinance will increase access to fresh and affordable food*, *News List*. Available at: https://www.atlantaga.gov/Home/Components/News/News/13829/ (Accessed: 24 January 2023).

City of Atlanta Mayor’s Office (2021b) *Substantial Increase in Fresh Food Access for Atlanta Residents*, *Press Releases*. Available at: https://www.atlantaga.gov/Home/Components/News/News/13777/672? (Accessed: 26 January 2023).

Clapp, J. (2017) ‘Food self-sufficiency: Making sense of it, and when it makes sense’, *Food Policy*, 66, pp. 88–96. doi:10.1016/j.foodpol.2016.12.001.

Clapp, J. (2021) ‘The problem with growing corporate concentration and power in the global food system’, *Nature Food*, 2(6), pp. 404–408. doi:10.1038/s43016-021-00297-7.

Clara, L. *et al.* (2017) *Soil organic carbon: the hidden potential*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://www.fao.org/3/i6937e/i6937e.pdf.

Clark, J.K., Conley, B. and Raja, S. (2021) 'Essential, fragile, and invisible community food infrastructure: The role of urban governments in the United States', *Food Policy*, 103(102014). doi:10.1016/j.foodpol.2020.102014.

Clayton, M.L. *et al.* (2015) 'The role of partnerships in U.S. Food policy council policy activities', *PLoS ONE*, 10(4), pp. 1–14. doi:10.1371/journal.pone.0122870.

Clemens, J. and Cuhls, C. (2003) 'Greenhouse gas emissions from mechanical and biological waste treatment of municipal waste', *Environmental Technology (United Kingdom)*, 24(6), pp. 745–754. doi:10.1080/09593330309385611.

Cleveland, D.A. *et al.* (2017) 'The potential for urban household vegetable gardens to reduce greenhouse gas emissions', *Landscape and Urban Planning*, 157, pp. 365–374. doi:10.1016/j.landurbplan.2016.07.008.

Clinton, N. *et al.* (2018) 'A Global Geospatial Ecosystem Services Estimate of Urban Agriculture', *Earth's Future*, 6. doi:10.1002/2017EF000536.

Coleman-Jensen, Alisha Rabbitt, M.P., Gregory, C.A. and Singh, A. (2022) *Household Food Security in the United States in 2021*. U.S. Department of Agriculture, Economic Research Service. Available at: https://www.ers.usda.gov/webdocs/publications/104656/err-309.pdf?v=869.2.

Colón, J. *et al.* (2009) 'Performance of an industrial biofilter from a composting plant in the removal of ammonia and VOCs after material replacement', *Journal of Chemical Technology and Biotechnology*, 84(8), pp. 1111–1117. doi:10.1002/jctb.2139.

Colón, J. *et al.* (2010) 'Environmental assessment of home composting', *Resources, Conservation and Recycling*, 54(11), pp. 893–904. doi:10.1016/j.resconrec.2010.01.008.

Committee on Climate Change (2017) *UK Climate Change Risk Assessment 2017*. London, UK. Available at: https://www.theccc.org.uk/wp-content/uploads/2016/07/UK-CCRA-2017-Synthesis-Report-Committee-on-Climate-Change.pdf.

Companioni, N., Rodríguez-Nodals, R. and Sardiñas, J. (2016) 'Capítulo 14: Agricultura urbana, suburbana y familiar', in Funes, F. and Vázquez, L. (eds) *Avances de la Agroecología en Cuba*. Matanzas, Cuba: Estación Experimental de Pastos y Forrajes Indio Hatuey, pp. 233–246.

Cooke, G.W. (1967) *The Control of Soil Fertility*. London, UK: Crosby Lockwood and Son.

Cooper, J. *et al.* (2018) 'Phosphorus availability on many organically managed farms in Europe', *Nutrient Cycling in Agroecosystems*, 110(2), pp. 227–239. doi:10.1007/s10705-017-9894-2.

Cooper, N., Purcell, S. and Jackson, R. (2014) *Below the Breadline: The Relentless Rise of Food Poverty in Britain*. Manchester, UK: Church Action on Poverty, Oxfam, and the Trussell Trust. Available at: https://www.trusselltrust.org/wp-content/uploads/sites/2/2016/01/Below-the-Breadline-The-Trussell-Trust.pdf.

Corbin, J. and Strauss, A. (2012) 'Basics of Qualitative Research: Techniques and Procedures for Developing Grounded Theory', in Corbin, J. and Strauss, A. (eds) *Basics of Qualitative Research*. 3rd edn. Thousand Oaks, CA, USA: SAGE Publications, Inc., pp. 1–18. doi:10.4135/9781452230153.

Costanza, R. *et al.* (1997) 'The value of the world's ecosystem services and natural capital', *Nature*, 387, pp. 253–260. doi:10.1038/387253a0.

Coucheney, E. *et al.* (2015) 'Accuracy, robustness and behavior of the STICS soil-crop model for plant, water and nitrogen outputs: Evaluation over a wide range of agro-environmental conditions in France', *Environmental Modelling and Software*, 64, pp. 177–190. doi:10.1016/j.envsoft.2014.11.024.

Cowan, K. (2020) *Survey results: Understanding people's concerns about the mental health impacts of the COVID-19 pandemic*. London, UK: The Academy of Medical Sciences and MQ: Transforming Mental Health. Available at: https://acmedsci.ac.uk/file-download/99436893.

Cox, C. and Surgan, M. (2006) 'Unidentified Inert Ingredients in Pesticides: Implications for Human and Environmental Health', *Environmental Health Perspectives*, 114(12), pp. 1803–1806. doi:10.1289/ehp.9374.

Cranfield University (2006) *World Reference Base Soil Map of England and Wales - NATMAP wrb, LandIS*. Available at: https://www.landis.org.uk/data/nmwrb.cfm (Accessed: 5 January 2023).

Cranfield University (2011) *Soilscapes of England and Wales*. Cranfield, UK. Available at: https://www.landis.org.uk/soilscapes/index.cfm.

Cranfield University (2023) *The Soils Guide: World Reference Base for England and Wales*, *LandIS*. Available at: https://www.landis.org.uk/soilsguide/wrb_list.cfm? (Accessed: 5 January 2023).

Crippa, M. *et al.* (2021) 'Food systems are responsible for a third of global anthropogenic GHG emissions', *Nature Food*, 2(3), pp. 198–209. doi:10.1038/s43016-021-00225-9.

Crispo, M. *et al.* (2021) 'Heavy metals and metalloids concentrations across UK urban horticultural soils and the factors influencing their bioavailability to food crops', *Environmental Pollution*, 288(117960). doi:10.1016/j.envpol.2021.117960.

Crouch, D. and Ward, C. (2003) *The allotment: its landscape and culture*. Nottingham, UK: Five Leaves Publications.

Cucinotta, D. and Vanelli, M. (2020) 'WHO Declares COVID-19 a Pandemic', *Acta Biomedica*, 91(1), pp. 157–60. doi:10.23750/abm.v91i1.9397.

Culbard, E.B. *et al.* (1988) 'Metal Contamination in British Urban Dusts and Soils', *Journal of Environment Quality*, 17(2), p. 226. doi:10.2134/jeq1988.00472425001700020011x.

Cullather, N. (2004) 'Miracles of Modernization: The Green Revolution and the Apotheosis of Technology', *Diplomatic History*, 28(2), pp. 227–254. doi:10.1111/j.1467-7709.2004.00407.x.

Cunningham, D.A., Collins, J.F. and Cummins, T. (2001) 'Anthropogenically-triggered iron pan formation in some Irish soils over various time spans', *Catena*, 43(3), pp. 167–176. doi:10.1016/S0341-8162(00)00161-2.

D'Odorico, P. *et al.* (2013) 'Global desertification: Drivers and feedbacks', *Advances in Water Resources*, 51, pp. 326–344. doi:10.1016/j.advwatres.2012.01.013.

Dabney, S.M. *et al.* (2010) 'Using cover crops and cropping systems for nitrogen management', in Delgado, J.A. and Follett, R.F. (eds) *Advances in Nitrogen Management for Water Quality*. Ankeny, IA, USA: U.S. Department of Agriculture, Agricultural Research Service, pp. 230–281. Available at: https://www.ars.usda.gov/research/publications/publication/?seqNo115=236261.

Daddow, R. and Warrington, G. (1983) *Growth-limiting soil bulk densities as influenced by soil texture*. Fort Collins, CO, USA: Watershed System Development Group, USDA Forest Service.

Daftary-Steel, S., Herrera, H. and Porter, C. (2015) 'The Unattainable Trifecta of Urban Agriculture', *Journal of Agriculture, Food Systems, and Community Development*, 6(1), pp. 19–32. doi:10.5304/jafscd.2015.061.014.

Daily, G.C. (1997) 'Introduction: What are Ecosystem Services?', in Daily, G.C. (ed.) *Nature's Services: Societal Dependence On Natural Ecosystems*. Washington DC: Island Press, pp. 1–10.

Daily, G.C., Matson, P.A. and Vitousek, P.M. (1997) 'Ecosystem Services Supplied by Soils', in Daily, G.C. (ed.) *Nature's Services: Societal Dependence On Natural Ecosystems*. Washington DC: Island Press, pp. 113–132.

Dalgaard, R. and Halberg, N. (2007) 'How to account for emissions from manure? Who bears the burden?', in *5th International Conference LCA in Foods*. Gothenburg, Sweden. Available at: http://lcafood.dk/Material/How to account.pdf.

Daly, M. (2020) 'COVID-19 and care homes in England: What happened and why?', *Social Policy & Administration*, 54(7), pp. 985–998. doi:10.1111/spol.12645.

Daly, M., Sutin, A.R. and Robinson, E. (2022) 'Longitudinal changes in mental health and the COVID-19 pandemic: evidence from the UK Household Longitudinal Study', *Psychological Medicine*, 52(13), pp. 2549–2558. doi:10.1017/S0033291720004432.

Dankwa-Mullan, I. and Pérez-Stable, E.J. (2016) 'Addressing Health Disparities Is a Place-Based Issue', *American Journal of Public Health*, 106(4), pp. 637–639. doi:10.2105/AJPH.2016.303077.

Danner, H. *et al.* (2022) 'The news media and its audience: Agenda setting on organic food in the United States and Germany', *Journal of Cleaner Production*, 354(131503). doi:10.1016/j.jclepro.2022.131503.

Darch, T. *et al.* (2014) 'A meta-analysis of organic and inorganic phosphorus in organic fertilizers, soils, and water: Implications for water quality', *Critical Reviews in Environmental Science and Technology*, 44(19), pp. 2172–2202. doi:10.1080/10643389.2013.790752.

Daum, D. and Schenk, M.K. (1996a) 'Gaseous nitrogen losses from a soilless culture system in the greenhouse', *Plant and Soil*, 183, pp. 69–78. doi:10.1007/BF02185566.

Daum, D. and Schenk, M.K. (1996b) 'Influence of nitrogen concentration and form in the nutrient solution N2O and N2 emissions from a soilless culture system', *Zeitschrift für Pflanzenernährung und Bodenkunde*, 159(6), pp. 557–563. doi:10.1002/jpln.1996.3581590606.

Daum, D. and Schenk, M.K. (1998) 'Influence of nutrient solution pH on N2O and N2 emissions from a soilless culture system', *Plant and Soil*, 203(2), pp. 279–288. doi:10.1023/A:1004350628266.

Davidson, E.A. (2009) 'The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860', *Nature Geoscience*, 2(9), pp. 659–662. doi:10.1038/ngeo608.

Davis, H.T. *et al.* (2009) 'Identifying natural and anthropogenic sources of metals in urban and rural soils using GIS-based data, PCA, and spatial interpolation', *Environmental Pollution*, 157(8–9), pp. 2378–2385. doi:10.1016/j.envpol.2009.03.021.

Defra (2009) *Safeguarding our Soils: A Strategy for England*. London, UK. Available at: https://www.gov.uk/government/publications/safeguarding-our-soils-a-strategy-for-england.

Defra (2011) *The Natural Choice: securing the value of nature*. London, UK. Available at: https://www.gov.uk/government/publications/the-natural-choice-securing-the-value-of-nature.

Defra (2012) *Environmental Protection Act 1990: Part 2A Contaminated Land Statutory Guidance*. London, UK. Available at: https://www.gov.uk/government/publications/contaminated-land-statutory-guidance.

Defra (2018) *Code of Good Agricultural Practice (COGAP) for Reducing Ammonia Emissions, Guidance*. Available at: https://www.gov.uk/government/publications/code-of-good-agricultural-practice-for-reducing-ammonia-emissions/code-of-good-agricultural-practice-cogap-for-reducing-ammonia-emissions (Accessed: 28 July 2022).

Defra (2019) *Horticulture Statistics Dataset 2018*. London. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/822083/hort-dataset-01aug19.xlsx.

Defra (2020) *The Path to Sustainable Farming: An Agricultural Transition Plan 2021 to 2024*. London, UK. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/954

283/agricultural-transition-plan.pdf.

Defra (2021a) *Soil Health Action Plan to be launched*, *Blog*. Available at: https://deframedia.blog.gov.uk/2021/09/09/soil-health-action-plan-to-be-launched/ (Accessed: 9 January 2023).

Defra (2021b) *United Kingdom Food Security Report 2021*. London, UK. Available at: https://www.gov.uk/government/statistics/united-kingdom-food-security-report-2021.

Defra (2021c) *Waste Management Plan for England*. London, UK: UK Department for Environment, Food and Rural Affairs. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/955897/waste-management-plan-for-england-2021.pdf.

Defra (2022a) *Agricultural land use in England at 1 June 2022*, *National statistics*. Available at: https://www.gov.uk/government/statistics/agricultural-land-use-in-england/agricultural-land-use-in-england-at-1-june-2022 (Accessed: 30 January 2023).

Defra (2022b) *Agriculture in the United Kingdom 2021*. London, UK. Available at: https://www.gov.uk/government/statistics/agriculture-in-the-united-kingdom-2021.

Defra (2022c) *Government food strategy*. London, UK. Available at: https://www.gov.uk/government/publications/government-food-strategy.

Defra (2022d) *New farming policies and payments in England*. London, UK. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1096744/payments-for-farmers.pdf.

Defra & Natural England (2022) *Delivering on the Environment Act: new targets announced and ambitious plans for nature recovery*, *Press Release*. Available at: https://www.gov.uk/government/news/delivering-on-the-environment-act-new-targets-announced-and-ambitious-plans-for-nature-recovery (Accessed: 9 January 2023).

Defra & Rural Payments Agency (2021) *Environmental Land Management schemes: overview*, *Guidance*. Available at: https://www.gov.uk/government/publications/environmental-land-management-schemes-overview/environmental-land-management-scheme-overview (Accessed: 7 January 2023).

Defra & Rural Payments Agency (2022a) *A summary of the SFI in 2022*, *Guidance*. Available at: https://www.gov.uk/guidance/a-summary-of-the-sfi-in-2022 (Accessed: 7 January 2023).

Defra & Rural Payments Agency (2022b) *Sustainable Farming Incentive: full guidance*. Available at: https://www.gov.uk/government/publications/sustainable-farming-incentive-full-guidance/sustainable-farming-incentive-full-guidance (Accessed: 27 January 2023).

Defra & Rural Payments Agency (2022c) *The SFI arable and horticultural soils standard*. Available at: https://www.gov.uk/guidance/the-sfi-arable-and-horticultural-soils-standard (Accessed: 7 January 2023).

Defra & Rural Payments Agency (2023) *Basic Payment Scheme (BPS)*. Available at: https://www.gov.uk/government/collections/basic-payment-scheme#bps-2023 (Accessed: 24 January 2023).

Dejong-Hughes, J. and Daigh, A. (2022) *Upper Midwest Soil Compaction Guide*. Minneapolis, MN, USA: University of Minnesota Extension. Available at: https://extension.umn.edu/soil-management-and-health/soil-compaction.

Dekker, E. *et al.* (2019) ‘A taste of the new ReCiPe for life cycle assessment: consequences of the updated impact assessment method on food product LCAs’, *The International Journal of Life Cycle Assessment*, 25(12), pp. 2315–2324. doi:10.1007/s11367-019-01653-3.

Demeter Association (2019) *Demeter Biodynamic Certification*. Available at: https://www.demeter-usa.org/certification/ (Accessed: 20 December 2019).

Demirbas, A. (2010) 'Oil, micronutrient and heavy metal contents of tomatoes', *Food Chemistry*, 118(3), pp. 504–507. doi:10.1016/j.foodchem.2009.05.007.

Dempsey, D. *et al.* (2021) *Food insecurity in times of Covid-19-an insight into a deepening crisis*. Glasgow, UK. Available at: https://oxfampartnership.uws.ac.uk/wp-content/uploads/2021/10/Food-insecurity-in-times-of-Covid-19-2021-WEB-FINAL.pdf.

Department for Work & Pensions (2021) *Universal Credit Statistics: 29 April 2013 to 8 July 2021, Official statistics*. Available at: https://www.gov.uk/government/statistics/universal-credit-statistics-29-april-2013-to-8-july-2021 (Accessed: 17 October 2022).

Department for Work and Pensions (2020) *DWP benefits statistics: August 2020, National statistics*. Available at: https://www.gov.uk/government/statistics/dwp-benefits-statistics-august-2020/dwp-benefits-statistics-august-2020 (Accessed: 20 September 2022).

Department of Health and Social Care (2021) *New TV advert urges public to stay at home to protect the NHS and save lives, Press Release*. Available at: https://www.gov.uk/government/news/new-tv-advert-urges-public-to-stay-at-home-to-protect-the-nhs-and-save-lives (Accessed: 10 August 2021).

Department of Trade & Industry (2005) *UK Energy in Brief July 2005*. London, UK. Available at: https://archive.uea.ac.uk/~e680/energy/pdf_files/energy_in_brief/Energy_in_Brief_2005_file10738.pdf.

Desaules, A. and Studer, K. (1993) *NABO: Nationales Beobachtungsnetz, Messresultate 1985-1991, Schriftenreihe Umwelt Nr. 200*. Bern, Switzerland: BUWAL.

Despommier, D. (2011) 'The vertical farm: controlled environment agriculture carried out in tall buildings would create greater food safety and security for large urban populations', *Journal of Consumer Protection and Food Safety*, 6, pp. 233–236. doi:10.1007/s00003-010-0654-3.

Desquilbet, M., Dorin, B. and Couvet, D. (2017) 'Land Sharing vs Land Sparing to Conserve Biodiversity: How Agricultural Markets Make the Difference', *Environmental Modeling & Assessment*, 22(3), pp. 185–200. doi:10.1007/s10666-016-9531-5.

Devotta, S. and Sicars, S. (2005) 'Refrigeration', in Calvo, E. and Elgizouli, I. (eds) *Safeguarding the Ozone Layer and the Global Climate System: Special Report of the Intergovernmental Panel on Climate Change*. IPCC. Available at: https://archive.ipcc.ch/pdf/special-reports/sroc/sroc04.pdf.

Diaconeasa, M.C. *et al.* (2022) 'Media Discourse on Sustainable Consumption in Europe', *Environmental Communication*, 16(3), pp. 352–370. doi:10.1080/17524032.2021.1999295.

Dias, G.M. *et al.* (2017) 'Life cycle perspectives on the sustainability of Ontario greenhouse tomato production: Benchmarking and improvement opportunities', *Journal of Cleaner Production*, 140, pp. 831–839. doi:10.1016/j.jclepro.2016.06.039.

Diepenbrock, W. (2012) 'Energy Balance in Crop Production', *Journal of Agricultural Science and Technology. B*, 2(5b), p. 527.

Dimbleby, H. (2021) *National Food Strategy: The Plan*. London, UK. doi:10.2307/j.ctt4cgqfz.3.

Dobson, M.C. *et al.* (2021) 'An assessment of urban horticultural soil quality in the United Kingdom and its contribution to carbon storage', *Science of the Total Environment*, 777. doi:10.1016/j.scitotenv.2021.146199.

Doherty, B. *et al.* (2019) 'Food Systems Resilience: Towards an Interdisciplinary Research Agenda', *Emerald Open Research*, 1(4). doi:10.12688/emeraldopenres.12850.1.

Dolan, V.L.B. (2021) '"…but if you tell anyone, I'll deny we ever met:" the experiences of academics with invisible disabilities in the neoliberal university', *International Journal of Qualitative Studies in Education* [Preprint]. doi:10.1080/09518398.2021.1885075.

Dominati, E., Patterson, M. and Mackay, A. (2010) 'A framework for classifying and quantifying the natural capital and ecosystem services of soils', *Ecological Economics*, 69(9), pp. 1858–1868. doi:10.1016/j.ecolecon.2010.05.002.

Donaher, E. and Lynes, J. (2017) 'Is local produce more expensive? Challenging perceptions of price in local food systems', *Local Environment*, 22(6), pp. 746–763. doi:10.1080/13549839.2016.1263940.

Dones, R. *et al.* (2007) *Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries*. Dübendorf, Switzerland: ecoinvent Centre. Available at: http://ecolo.org/documents/documents_in_english/Life-cycle-analysis-PSI-05.pdf.

Doran, J.W. and Safley, M. (1997) 'Defining and assessing soil health and sustainable productivity', in Pankhurst, C.E., Doube, B.M., and Gupta, V.V. (eds) *Biological Indicators of Soil Health*. Wallingford, UK: CAB International, pp. 1–28.

Dorr, E. *et al.* (2021) 'Environmental impacts and resource use of urban agriculture: a systematic review and meta-analysis', *Environmental Research Letters*, 16(093002). doi:10.1088/1748-9326/ac1a39.

Douglas, F. *et al.* (2015) 'Resourcefulness, Desperation, Shame, Gratitude and Powerlessness: Common Themes Emerging from A Study of Food Bank Use in Northeast Scotland', *AIMS Public Health*, 2(3), pp. 297–317. doi:10.3934/publichealth.2015.3.297.

Dowler, E. (2002) 'Food and Poverty in Britain: Rights and Responsibilities', *Social Policy and Administration*, 36(6), pp. 698–717. doi:10.1111/1467-9515.00312.

Dowler, E. *et al.* (2009) '"Doing food differently": Reconnecting biological and social relationships through care for food', *Sociological Review*, 57(Suppl 2), pp. 200–221. doi:10.1111/j.1467-954X.2010.01893.x.

Dowler, E., Caraher, M. and Lincoln, P. (2007) 'Inequalities in food and nutrition: challenging "lifestyles"', in Dowler, E. and Spencer, N. (eds) *Challenging Health Inequalities: From Acheson to 'Choosing Health'*. Bristol, UK: The Policy Press, pp. 127–155.

Dowler, E. and Lambie-Mumford, H. (2015) 'How can households eat in austerity? Challenges for social policy in the UK', *Social Policy and Society*, 14(3), pp. 417–428. doi:10.1017/S1474746415000032.

Dowler, E., Turner, S. and Dobson, B. (2001) *Poverty Bites: Food, Health and Poor Families*. London: Child Poverty Action Group.

Dowler, E.A. and O'Connor, D. (2012) 'Rights-based approaches to addressing food poverty and food insecurity in Ireland and UK', *Social Science & Medicine*, 74(1), pp. 44–51. doi:10.1016/j.socscimed.2011.08.036.

Drury, J. *et al.* (2019) 'Facilitating Collective Psychosocial Resilience in the Public in Emergencies: Twelve Recommendations Based on the Social Identity Approach', *Frontiers in Public Health*, 7. doi:10.3389/fpubh.2019.00141.

Dunbar, R.I.M. (1998) 'The social brain hypothesis', *Evolutionary Anthropology*, 6(5), pp. 178–190. doi:10.1002/(SICI)1520-6505(1998)6:5<178::AID-EVAN5>3.0.CO;2-8.

Dunbar, R.I.M. (2017) 'Breaking Bread: the Functions of Social Eating', *Adaptive Human Behavior and Physiology*, 3(3), pp. 198–211. doi:10.1007/s40750-017-0061-4.

Dungait, J.A.J. *et al.* (2012) 'Advances in the understanding of nutrient dynamics and management in UK agriculture', *Science of the Total Environment*, 434, pp. 39–50. doi:10.1016/j.scitotenv.2012.04.029.

Dupraz, P. and Guyomard, H. (2019) ‘Environment and Climate in the Common Agricultural Policy’, *EuroChoices*, 18(1), pp. 18–25. doi:10.1111/1746-692X.12219.

DuPuis, E.M., Ransom, E. and Worosz, M.R. (2022) ‘Food Supply Chain Shocks and the Pivot Toward Local: Lessons From the Global Pandemic’, *Frontiers in Sustainable Food Systems*, 6(836574). doi:10.3389/fsufs.2022.836574.

Durlinger, B. *et al.* (2017) *Agri-footprint 4.0. Part 2: Description of data*. Gouda, NL: Blonk Consultants.

Dyer, C. (2020) ‘Covid-19: Coroners needn’t investigate PPE policy failures in deaths of NHS staff, new guidance says’, *BMJ*, p. m1806. doi:10.1136/bmj.m1806.

Dyer, J.A. *et al.* (2011) ‘Comparing fossil CO2 emissions from vegetable greenhouses in Canada with CO2 emissions from importing vegetables from the southern USA’, *Energy for Sustainable Development*, 15(4), pp. 451–459. doi:10.1016/j.esd.2011.08.004.

Eastburn, D.J. *et al.* (2017) ‘Multiple ecosystem services in a working landscape’, *PLOS ONE*, 12(3), p. e0166595. doi:10.1371/journal.pone.0166595.

ecoinvent Centre (2021) *ecoinvent data v.3.7.1*. St. Gallen, Switzerland. Available at: www.ecoinvent.org.

Economic Affairs Committee (2020) *Universal Credit isn’t working: proposals for reform*. London, UK: House of Lords, UK Parliament. Available at: https://publications.parliament.uk/pa/ld5801/ldselect/ldeconaf/105/105.pdf.

Eddleston, M. *et al.* (2012) ‘A role for solvents in the toxicity of agricultural organophosphorus pesticides’, *Toxicology*, 294(2–3), pp. 94–103. doi:10.1016/j.tox.2012.02.005.

Edmondson, J.L. *et al.* (2011) ‘Are soils in urban ecosystems compacted? A citywide analysis’, *Biology Letters*, 7(5), pp. 771–774. doi:10.1098/rsbl.2011.0260.

Edmondson, J.L. *et al.* (2012) ‘Organic carbon hidden in urban ecosystems’, *Scientific Reports*, 2, pp. 1–7. doi:10.1038/srep00963.

Edmondson, J.L. *et al.* (2014) ‘Urban cultivation in allotments maintains soil qualities adversely affected by conventional agriculture’, *Journal of Applied Ecology*, 51, pp. 880–889. doi:10.1111/1365-2664.12254.

Edmondson, J.L. *et al.* (2020) ‘The hidden potential of urban horticulture’, *Nature Food*, 1, pp. 155–159. doi:10.1038/s43016-020-0045-6.

Edwards, A.R. *et al.* (1978) ‘The potential of sunflower as a crop for ensilage’, *Journal of the Science of Food and Agriculture*, 29(4), pp. 332–338. doi:10.1002/jsfa.2740290406.

EFRA (2020a) *COVID-19 and Food Supply: First Report of Session 2019–21*. London: House of Commons, Environment Food and Rural Affairs Committee. Available at: https://committees.parliament.uk/publications/2187/documents/20156/default/.

EFRA (2020b) *COVID-19 and food supply*, *Inquiry: Environment, Food and Rural Affairs Committee*. Available at: https://committees.parliament.uk/work/217/covid19-and-food-supply/ (Accessed: 13 September 2021).

EFRA (2021) *COVID-19 and food supply: follow up*, *Inquiry: Environment, Food and Rural Affairs Committee*. Available at: https://committees.parliament.uk/work/959/covid19-and-food-supply-follow-up/publications/ (Accessed: 13 September 2022).

Ekvall, T. and Tillman, A.M. (1997) ‘Open-loop recycling: Criteria for allocation procedures’, *International Journal of Life Cycle Assessment*, 2(3), pp. 155–162. doi:10.1007/BF02978810.

Elgar, F.J., Stefaniak, A. and Wohl, M.J.A. (2020) ‘The trouble with trust: Time-series analysis of social capital, income inequality, and COVID-19 deaths in 84 countries’, *Social Science & Medicine*, 263(113365). doi:10.1016/j.socscimed.2020.113365.

Elliott, J. *et al.* (2021) ‘COVID-19 mortality in the UK Biobank cohort: revisiting and evaluating risk factors’, *European Journal of Epidemiology*, 36(3), pp. 299–309. doi:10.1007/s10654-021-00722-y.

Ellis, A. *et al.* (2021) ‘A ticking time bomb of future harm: Lockdown, child abuse and future violence’, *Abuse: An International Impact Journal*, 2(1), pp. 37–48. doi:10.37576/abuse.2021.017.

Elsa, C. and Desmond, G.M. (2021) ‘Vermicompost soil amendment influences yield, growth responses and nutritional value of Kale (Brassica oleracea Acephala group), Radish (Raphanus sativus) and Tomato (Solanum lycopersicum L)’, *Journal of Soil Science and Environmental Management*, 12(2), pp. 86–93. doi:10.5897/jssem2021.0873.

EMEP & EEA (2019a) ‘Crop production and agricultural soils’, in *EMEP/EEA air pollutant emission inventory guidebook 2019 - Technical guidance to prepare national emission inventories*. Luxembourg: European Environment Agency. Available at: https://www.eea.europa.eu/publications/emep-eea-guidebook-2019.

EMEP & EEA (2019b) ‘Manure management’, in *EMEP/EEA air pollutant emission inventory guidebook 2019 - Technical guidance to prepare national emission inventories*. Luxembourg: European Environment Agency. Available at: https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/4-agriculture/3-b-manure-management/view.

EMEP & EEA (2019c) ‘Small combustion’, in *EMEP/EEA air pollutant emission inventory guidebook 2019 - Technical guidance to prepare national emission inventories*. Luxembourg: European Environment Agency. Available at: https://www.eea.europa.eu/publications/emep-eea-guidebook-2019.

Emmenegger, M.F., Reinhard, J. and Zah, R. (2009) *Sustainability Quick Check for Biofuels - Background Report*. Dübendorf, Switzerland. Available at: http://rsb.epfl.ch/files/content/sites/rsb2/files/Biofuels/Working Groups/GHG EG/SQCB_Background_report_en.pdf.

Emmerson, M. *et al.* (2016) ‘How Agricultural Intensification Affects Biodiversity and Ecosystem Services’, in Dumbrell, A.J., Kordas, R.L., and Woodward, G. (eds) *Advances in Ecological Research*. London, UK: Elsevier, pp. 43–97. doi:10.1016/bs.aecr.2016.08.005.

Emmett, B.A. *et al.* (2010) *Countryside Survey: Soils Report from 2007*. Wallingford, UK. Available at: https://countrysidesurvey.org.uk/sites/default/files/CS_UK_2007_TR9-revised - Soils Report.pdf.

Entwistle, J.A. *et al.* (2019) ‘An apple a day? Assessing gardeners’ lead exposure in urban agriculture sites to improve the derivation of soil assessment criteria’, *Environment International*, 122, pp. 130–141. doi:10.1016/j.envint.2018.10.054.

Environment Agency (2022) ‘All of England’s South West region now in drought’, *Press Release*, 30 August. Available at: https://www.gov.uk/government/news/all-of-england-s-south-west-region-now-in-drought.

Esteves, R.C., Vendramini, A.L. do A. and Accioly, F. (2021) ‘A qualitative meta-synthesis study of the convergence between organic crop regulations in the United States, Brazil, and Europe’, *Trends in Food Science and Technology*, 107(November 2020), pp. 343–357. doi:10.1016/j.tifs.2020.10.044.

European Commission (2010) *International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance*. 1st edn. Luxembourg: Publications Office of the European Union.

European Commission (2014) *Support for rural development by the European Agricultural Fund for Rural Development (807/ 2014; Commission Delegated Regulation (EU))*.

European Commission (2018) *Product Environmental Footprint Category Rules Guidance (PEFCRs)*. Available at: https://ec.europa.eu/environment/eussd/smgp/pdf/PEFCR_guidance_v6.3.pdf.

European Commission (2020) *A Farm to Fork Strategy for a fair, healthy and environmentally-friendly food system*. Brussels, Belgium. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52020DC0381.

European Commission (2022) *The Environmental Footprint Pilots*. Available at: https://ec.europa.eu/environment/eussd/smgp/ef_pilots.htm (Accessed: 16 December 2022).

European Commission (2023) *The common agricultural policy at a glance*, *Agriculture and rural development*. Available at: https://agriculture.ec.europa.eu/common-agricultural-policy/cap-overview/cap-glance_en (Accessed: 26 January 2023).

European Court of Auditors (2017) *Greening: a more complex income support scheme, not yet environmentally effective*. Luxembourg. Available at: https://www.eca.europa.eu/Lists/ECADocuments/SR17_21/SR_GREENING_EN.pdf.

European Soil Bureau (2001) *Georeferenced Soil Database for Europe: Manual of Procedures Version 1.1. EUR 18092 EN.* Luxembourg. Available at: https://esdac.jrc.ec.europa.eu/content/georeferenced-soil-database-europe-manual-procedures-version-11.

Fabian, E.E. and Smith-Zajaczkowski, J. (2019) *Horse Stable Maure Management*. University Park, PA: Pennsylvania State University Extension. Available at: https://extension.psu.edu/horse-stable-manure-management.

Falkenmark, M. (2013) ‘Growing water scarcity in agriculture: future challenge to global water security’, *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 371(2002), p. 20120410. doi:10.1098/rsta.2012.0410.

FAO (1999) ‘Organic Agriculture’, in *Committee on Agriculture, 15th Session, Item 8 of the Provisional Agenda*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://www.fao.org/3/X0075e/X0075e.htm.

FAO (2007) *Profitability and sustainability of urban and peri-urban agriculture. Agricultural Management, Marketing and Finance Occasional Paper 19*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://ruaf.org/assets/2019/11/Profitability-and-Sustainability.pdf.

FAO (2008) *An Introduction to the Basic Concepts of Food Security*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: http://www.fao.org/3/al936e/al936e00.pdf (Accessed: 15 September 2022).

FAO (2009) *Global agriculture towards 2050*. Rome, Italy: Food and Agriculture Organization of the United Nations. doi:10.5822/978-1-61091-885-5.

FAO (2011) *Organic Agriculture and Climate Change Mitigation: A Report of the Round Table on Organic Agriculture and Climate Change*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://www.fao.org/3/i2537e/i2537e00.pdf.

FAO (2015) *Healthy soils are the basis for healthy food production*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: http://www.fao.org/3/a-i4405e.pdf.

FAO (2017) *The future of food and agriculture: Trends and challenges*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: http://www.fao.org/3/a-i6583e.pdf.

FAO (2018) *The 10 elements of agroecology: Guiding the transition to sustainable food and agricultural systems*. Rome: Food and Agriculture Organization of the United Nations. Available at:

http://www.fao.org/3/I9037EN/i9037en.pdf.

FAO (2021) *The state of the world's land and water resources for food and agriculture: Systems at a breaking point*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://www.fao.org/3/cb7654en/cb7654en.pdf.

FAO *et al.* (2022) *The State of Food Security and Nutrition in the World 2022. Repurposing food and agricultural policies to make healthy diets more affordable*. Rome, Italy: Food and Agriculture Organization of the United Nations. Available at: https://www.fao.org/publications/sofi/2022/en/.

FAO & WHO (2007) *CODEX Alimentarius: Organically Produced Foods*. Rome, Italy: Food and Agriculture Organization and World Health Organization. Available at: https://www.fao.org/3/a1385e/a1385e00.pdf.

FAO and ITPS (2015) *Status of the World's Soil Resources (SWSR) - Main Report*. Rome, Italy: Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils. Available at: http://www.fao.org/3/i5199e/I5199E.pdf.

Farago, M.E. *et al.* (1997) 'Health aspects of human exposure to high arsenic concentrations in soil in south-west England', in Abernathy, C.O., Calderon, R.L., and Chappell, W.R. (eds) *Arsenic*. Dordrecht, Netherlands: Springer, pp. 210–226. doi:10.1007/978-94-011-5864-0_17.

Faris, J.E. (1961) 'Economies of Scale in Crop Production', *Journal of Farm Economics*, 43(5), p. 1219. doi:10.2307/1235579.

FarmingUK (2022) 'UK farming faces "talent drought" as young people steer away', *Education, News, Rural Life*, 30 June. Available at: https://www.farminguk.com/news/uk-farming-faces-talent-drought-as-young-people-steer-away_60668.html.

Farooq, M.S. *et al.* (2022) 'Uncovering the Research Gaps to Alleviate the Negative Impacts of Climate Change on Food Security: A Review', *Frontiers in Plant Science*, 13(927535). doi:10.3389/fpls.2022.927535.

Fei, S., Ni, J. and Santini, G. (2020) 'Local food systems and COVID-19: an insight from China', *Resources, Conservation and Recycling*, 162(105022). doi:10.1016/j.resconrec.2020.105022.

Feldmann, C. and Hamm, U. (2015) 'Consumers' perceptions and preferences for local food: A review', *Food Quality and Preference*, 40(Part A), pp. 152–164. doi:10.1016/j.foodqual.2014.09.014.

Feniuk, C., Balmford, A. and Green, R.E. (2019) 'Land sparing to make space for species dependent on natural habitats and high nature value farmland', *Proceedings of the Royal Society B: Biological Sciences*, 286(1909). doi:10.1098/rspb.2019.1483.

Fernandes-Jesus, M. *et al.* (2021) 'More Than a COVID-19 Response: Sustaining Mutual Aid Groups During and Beyond the Pandemic', *Frontiers in Psychology*, 12(716202). doi:10.3389/fpsyg.2021.716202.

Fernandez, M. *et al.* (2018) 'New opportunities, new challenges: Harnessing Cuba's advances in agroecology and sustainable agriculture in the context of changing relations with the United States', *Elementa Science of the Anthropocene*, 6(76). doi:10.1525/elementa.337.

Filimonau, V. (2021) 'The prospects of waste management in the hospitality sector post COVID-19', *Resources, Conservation and Recycling*, 168(105272). doi:10.1016/j.resconrec.2020.105272.

Finch, T. *et al.* (2019) 'Bird conservation and the land sharing-sparing continuum in farmland-dominated landscapes of lowland England', *Conservation Biology*, 33(5), pp. 1045–1055. doi:10.1111/cobi.13316.

Fine, A.K., van Es, H.M. and Schindelbeck, R.R. (2017) 'Statistics, Scoring Functions, and Regional Analysis of a Comprehensive Soil Health Database', *Soil Science Society of America Journal*, 81(3), pp. 589–601. doi:10.2136/sssaj2016.09.0286.

Finnveden, G. *et al.* (2009) 'Recent developments in Life Cycle Assessment', *Journal of Environmental Management*, 91, pp. 1–21. doi:10.1016/j.jenvman.2009.06.018.

Firth, R. (2020) 'Mutual aid, anarchist preparedness and COVID-19', in Preston, J. and Firth, R. (eds) *Coronavirus, Class and Mutual Aid in the United Kingdom*. London, UK: Palgrave Macmillan, pp. 57–112.

Fischer, J. *et al.* (2011) 'Conservation: Limits of Land Sparing', *Science*, 334(6056), pp. 593–593. doi:10.1126/science.334.6056.593-a.

Fischler, C. (2011) 'Commensality, society and culture', *Social Science Information*, 50(3–4), pp. 528–548. doi:10.1177/0539018411413963.

Foley, J.A. *et al.* (2011) 'Solutions for a cultivated planet', *Nature*, 478(7369), pp. 337–342. doi:10.1038/nature10452.

Food Well Alliance (2017) *Atlanta's Local Food Baseline Report*. Available at: http://www.foodwellalliance.org/baselinereport.

Fortune, S. *et al.* (2005) 'Assessment of phosphorus leaching losses from arable land', *Plant and Soil*, 269(1–2), pp. 99–108. doi:10.1007/s11104-004-1659-4.

Foteinis, S. and Chatzisymeon, E. (2016) 'Life cycle assessment of organic versus conventional agriculture. A case study of lettuce cultivation in Greece', *Journal of Cleaner Production*, 112, pp. 2462–2471. doi:10.1016/j.jclepro.2015.09.075.

Foucault, M. (2008) *The Birth of Biopolitics*. Basingstoke, UK: Palgrave.

Francis-Devine, B., Powell, A. and Clark, H. (2021) *Coronavirus Job Retention Scheme: statistics*. House of Commons Library. Available at: https://commonslibrary.parliament.uk/research-briefings/cbp-9152/.

Frank, M., Laginess, T. and Schöneboom, J. (2020) 'Social Life Cycle Assessment in Agricultural Systems – U.S. Corn Production as a Case Study', in Traverso, M., Petti, L., and Zamagni, A. (eds) *Perspectives on Social LCA*. Cham, Switzerland: Springer, pp. 119–129. doi:10.1007/978-3-030-01508-4_11.

Frankowska, A., Jeswani, H.K. and Azapagic, A. (2019) 'Environmental impacts of vegetables consumption in the UK', *Science of the Total Environment*, 682, pp. 80–105. doi:10.1016/j.scitotenv.2019.04.424.

Freiermuth, R. (2006) *Modell zur Berechnung der Schwermetallflüsse in der Landwirtschaftlichen Ökobilanz*. Zurich, Switzerland: Agroscope FAL Reckenholz. Available at: https://www.agroscope.admin.ch/dam/agroscope/de/dokumente/themen/umwelt-ressourcen/produktionssysteme/salca-schwermetall.pdf.download.pdf/SALCA-Schwermetall.pdf.

Frick, H. *et al.* (2022) 'Leached nitrate under fertilised loamy soil originates mainly from mineralisation of soil organic N', *Agriculture, Ecosystems & Environment*, 338(108093). doi:10.1016/j.agee.2022.108093.

Gabriel, K.R. (1971) 'The Biplot Graphic Display of Matrices with Application to Principal Component Analysis', *Biometrika*, 58(3), p. 453. doi:10.2307/2334381.

Gamero, A. *et al.* (2017) 'Tracking Progress Toward EU Biodiversity Strategy Targets: EU Policy Effects in Preserving its Common Farmland Birds', *Conservation Letters*, 10(4), pp. 395–402. doi:10.1111/conl.12292.

Garcia, E. (2019) 'Where's the waste? A circular economy could combat climate change', *The New York Times*, 21 September. Available at: https://www.nytimes.com/2019/09/21/climate/circular-food-economy-sustainable.html.

Gardner, W.H. (1986) ‘Water Content’, in Klute, A. (ed.) *Methods of Soil Analysis: Part 1 Physical and Mineralogical Methods*. 2nd edn. American Society of Agronomy, Inc. Soil Science Society of America, Inc., pp. 493–544. doi:10.2136/sssabookser5.1.2ed.c21.

Garnett, P., Doherty, B. and Heron, T. (2020) ‘Vulnerability of the United Kingdom’s food supply chains exposed by COVID-19’, *Nature Food*, 1(6), pp. 315–318. doi:10.1038/s43016-020-0097-7.

Garnett, T. (2011) ‘Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)?’, *Food Policy*, 36, pp. S23–S32. doi:10.1016/j.foodpol.2010.10.010.

Garnett, T. (2014) ‘Three perspectives on sustainable food security: Efficiency, demand restraint, food system transformation. What role for life cycle assessment?’, *Journal of Cleaner Production*, 73, pp. 10–18. doi:10.1016/j.jclepro.2013.07.045.

Garthwaite, K. (2016) *Hunger pains: Life inside foodbank Britain*. Bristol, UK: Policy Press.

Garthwaite, K.A., Collins, P.J. and Bambra, C. (2015) ‘Food for thought: An ethnographic study of negotiating ill health and food insecurity in a UK foodbank’, *Social Science and Medicine*, 132, pp. 38–44. doi:10.1016/j.socscimed.2015.03.019.

Gattinger, A. *et al.* (2012) ‘Enhanced top soil carbon stocks under organic farming’, *Proceedings of the National Academy of Sciences of the United States of America*, 109(44), pp. 18226–18231. doi:10.1073/pnas.1209429109.

Georgia Environmental Protection Division (2021) *Regulated Solid Waste Facilities*. Available at: https://epd.georgia.gov/about-us/land-protection-branch/solid-waste/regulated-solid-waste-facilities (Accessed: 18 July 2022).

GFS (2019) *Exploring the resilience of the UK food system in a global context*. Oxford, UK: Global Food Security Programme. Available at: https://www.foodsecurity.ac.uk/wp-content/uploads/2009/10/exploring-the-resilience-of-the-uk-food-system-in-a-global-context.pdf.

Ghate, D. and Hazel, N. (2002) *Parenting in poor environments: Stress, support and coping*. London, UK: Jessica Kingsley Publishers.

Gibson, J. (2020) ‘Domestic violence during COVID-19: the GP role’, *British Journal of General Practice*, 70(696), pp. 340–340. doi:10.3399/bjgp20X710477.

Gibson, M.J. and Farmer, J.G. (1986) ‘Multi-step sequential chemical extraction of heavy metals from urban soils’, *Environmental Pollution Series B, Chemical and Physical*, 11(2), pp. 117–135. doi:10.1016/0143-148X(86)90039-X.

Giebel, C. *et al.* (2021) ‘A UK survey of COVID-19 related social support closures and their effects on older people, people with dementia, and carers’, *International Journal of Geriatric Psychiatry*, 36(3), pp. 393–402. doi:10.1002/gps.5434.

Giusti, L. (2011) ‘Heavy metals in urban soils of Bristol (UK). Initial screening for contaminated land’, *Journal of Soils and Sediments*, 11(8), pp. 1385–1398. doi:10.1007/s11368-011-0434-4.

Godfray, H.C.J. *et al.* (2010) ‘The Challenge of Food Security’, *Science*, 327, pp. 812–818. doi:10.4337/9780857939388.

Goedkoop, M. *et al.* (2009) *ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition. Report I: Characterisation*. Netherlands. Available at: http://www.pre-sustainability.com/download/misc/ReCiPe_main_report_final_27-02-2009_web.pdf.

Goffman, I. (1963) *Stigma: Notes on the management of spoiled identity*. Englewood Cliffs, NJ, USA: Prentice-Hall.

Goglio, P. *et al.* (2018) ‘Development of Crop.LCA, an adaptable screening life cycle assessment tool for agricultural systems: A Canadian scenario assessment’, *Journal of Cleaner Production*, 172, pp. 3770–3780. doi:10.1016/j.jclepro.2017.06.175.

Goldin, I. and Winters, A.L. (1995) *The Economics of Sustainable Development*. Edited by Ian Goldin and L.A. Winters. Cambridge, UK: Cambridge University Press. doi:10.1017/CBO9780511751905.

Goldstein, B. *et al.* (2016) ‘Urban versus conventional agriculture, taxonomy of resource profiles: a review’, *Agronomy for Sustainable Development*, 36(9). doi:10.1007/s13593-015-0348-4.

Goldstein, N. (2009) ‘Vacant lots sprout urban farms’, *BioCycle*, 50(10), pp. 24–26.

Gomiero, T. (2016) ‘Soil degradation, land scarcity and food security: Reviewing a complex challenge’, *Sustainability*, 8(281). doi:10.3390/su8030281.

Gonzalez-Fernandez, O. *et al.* (2011) ‘Elemental characterization of edible plants and soils in an abandoned mining region: assessment of environmental risk’, *X-Ray Spectrometry*, 40(5), pp. 353–363. doi:10.1002/xrs.1348.

Goodley, D. (2014) *Dis/Ability Studies: Theorising Disablism and Ableism*. London, UK: Routledge.

Goodley, D. and Lawthom, R. (2019) ‘Critical disability studies, Brexit and Trump: a time of neoliberal–ableism’, *Rethinking History*, 23(2), pp. 233–251. doi:10.1080/13642529.2019.1607476.

Goodley, D., Lawthom, R. and Runswick-Cole, K. (2014) ‘Dis/ability and austerity: beyond work and slow death’, *Disability & Society*, 29(6), pp. 980–984. doi:10.1080/09687599.2014.920125.

Goodwin, H.L. (2005) ‘Location of Production and Consolidation in the Processing Industry: The Case of Poultry’, *Journal of Agricultural and Applied Economics*, 37(2), pp. 339–346. doi:10.1017/S1074070800006829.

Goodwin, S. (2022) ‘Food aid charities fear the worst as the cost of living crisis takes hold’, *BMJ*, 376, p. o416. doi:10.1136/bmj.o416.

Google (2022) ‘Google Earth Pro Version 7.3.6’. Available at: https://www.google.com/earth/versions/.

Gosling, P. and Shepherd, M. (2005) ‘Long-term changes in soil fertility in organic arable farming systems in England, with particular reference to phosphorus and potassium’, *Agriculture, Ecosystems & Environment*, 105(1–2), pp. 425–432. doi:10.1016/j.agee.2004.03.007.

Goucher, L. *et al.* (2017) ‘The environmental impact of fertilizer embodied in a wheat-to-bread supply chain’, *Nature Plants*, 3(17012). doi:10.1038/nplants.2017.12.

Goulding, K., Jarvis, S. and Whitmore, A. (2008) ‘Optimizing nutrient management for farm systems’, *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1491), pp. 667–680. doi:10.1098/rstb.2007.2177.

Goulding, K., Stockdale, E. and Watson, C. (2008) ‘Plant Nutrients in Organic Farming’, in Kirchmann, H. and Bergström, L. (eds) *Organic Crop Production - Ambitions and Limitations*. Springer, pp. 73–88.

Graddy-Lovelace, G. and Diamond, A. (2017) ‘From supply management to agricultural subsidies—and back again? The U.S. Farm Bill & agrarian (in)viability’, *Journal of Rural Studies*, 50, pp. 70–83. doi:10.1016/j.jrurstud.2016.12.007.

Granovetter, M.S. (1973) ‘The strength of weak ties’, *American Journal of Sociology*, 78(6), pp. 1360–1380.

GraphPad Software (2022) ‘GraphPad Prism for Windows: Version 9.5.0’. San Diego, California USA. Available at: www.graphpad.com.

Grasselly, D., Trédan, M. and Colomb, V. (2018) *AGRIBALYSE® fruit and vegetables: additions to the life cycle inventory database and the design of sustainable agricultural systems*. ADEME and CTIFL. Available at: https://librairie.ademe.fr/produire-autrement/101-agribalyse-fruit-and-vegetables.html.

Graven, C. *et al.* (2021) *The range and accessibility of food aid provision in Bradford, and the impact of COVID-19*. Bradford, UK: UK Prevention Research Partnership, University of York, and Bradford Institute for Health Research. Available at: https://www.bradfordresearch.nhs.uk/wp-content/uploads/2021/01/The-impact-of-COVID-19-on-the-provision-of-food-aid-in-Bradford_V4-Jan-21.pdf.

Graves, A. *et al.* (2011) *Cost of soil degradation in England and Wales: Defra research project final report*. Cranfield: UK Department for Environment, Food, and Rural Affairs & Cranfield University.

Greater London Authority (2018) *The London Food Strategy*. London, UK. Available at: https://www.london.gov.uk/sites/default/files/final_london_food_strategy.pdf.

Green Alliance (2022) *The opportunities of agri-carbon markets: a summary*. London, UK: Green Alliance UK. Available at: https://green-alliance.org.uk/wp-content/uploads/2022/01/The_opportunities_of_agri-carbon_markets_summary.pdf.

Green, M.B. (1987) ‘Energy in pesticide manufacture, distribution and use’, in Helsel, Z.. . (ed.) *Energy in plant nutrition and pest control*. Amsterdam: Elsevier, pp. 165–177.

Green, R.E. *et al.* (2005) ‘Farming and the Fate of Wild Nature’, *Science*, 307(5709), pp. 550–555. doi:10.1126/science.1106049.

Greenblatt, M., Becerra, R.M. and Serafetinides, E.A. (1982) ‘Social networks and mental health: on overview’, *American Journal of Psychiatry*, 139(8), pp. 977–984. doi:10.1176/ajp.139.8.977.

Gregory, A.S. *et al.* (2011) *Review of the evidence base for the status and change of soil carbon below 15cm from the soil surface in England and Wales*. UK Department for Environment, Food & Rural Affairs. Available at: https://repository.rothamsted.ac.uk/item/8q901.

Grisso, R. *et al.* (2014) *Predicting Tractor Fuel Diesel Consumption: Publication 442-073*. Blacksburg, VA: Virginia Cooperative Extension. doi:10.13031/2013.17455.

Groarke, J.M. *et al.* (2020) ‘Loneliness in the UK during the COVID-19 pandemic: Cross-sectional results from the COVID-19 Psychological Wellbeing Study’, *PLoS ONE*, 15(9 September), pp. 1–18. doi:10.1371/journal.pone.0239698.

Groenbaek, M. *et al.* (2014) ‘Influence of cultivar and fertilizer approach on curly kale (Brassica oleracea L. var. sabellica). 1. Genetic diversity reflected in agronomic characteristics and phytochemical concentration’, *Journal of Agricultural and Food Chemistry*, 62(47), pp. 11393–11402. doi:10.1021/jf503096p.

de Groot, R. (1992) *Functions of Nature: Evaluation of Nature in Environmental Planning, Management and Decision Making*. Groningen: Wolters-Noordhof.

Guerci, M. *et al.* (2013) ‘Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy’, *Journal of Cleaner Production*, 54, pp. 133–141. doi:10.1016/j.jclepro.2013.04.035.

Gundersen, C. and Ziliak, J.P. (2015) ‘Food Insecurity And Health Outcomes’, *Health Affairs*, 34(11), pp. 1830–1839. doi:10.1377/hlthaff.2015.0645.

Guo, L.B. and Gifford, R.M. (2002) ‘Soil carbon stocks and land use change: a meta analysis’, *Global Change Biology*, 8, pp. 345–360. doi:10.1046/j.1354-1013.2002.00486.x.

Hacker, K.A. *et al.* (2021) ‘COVID-19 and Chronic Disease: The Impact Now and in the Future’, *Preventing Chronic Disease*, 18(210086). doi:10.5888/pcd18.210086.

Haddaway, N.R. *et al.* (2017) ‘How does tillage intensity affect soil organic carbon? A systematic review’, *Environmental Evidence*, 6(1), p. 30. doi:10.1186/s13750-017-0108-9.

Hall, S.M. and Holmes, H. (2020) ‘Introduction: Mundane methods and the extra-ordinary everyday’, in Holmes, H. and Hall, S.M. (eds) *Mundane Methods*. Manchester: Manchester University Press, pp. 1–14.

Halliday, J. (2019) ‘Cities’ Strategies for Sustainable Food and the Levers They Mobilize’, in Brand, C. et al. (eds) *Designing Urban Food Policies. Urban Agriculture.* Cham, Switzerland: Springer, pp. 53–74. doi:10.1007/978-3-030-13958-2_3.

Hamelin, A.M., Beaudry, M. and Habicht, J.P. (2002) ‘Characterization of household food insecurity in Québec: Food and feelings’, *Social Science and Medicine*, 54(1), pp. 119–132. doi:10.1016/S0277-9536(01)00013-2.

Hamelin, A.M., Mercier, C. and Bedard, A. (2010) ‘Discrepancies in households and other stakeholders viewpoints on the food security experience: a gap to address’, *Health Education Research*, 25(3), pp. 401–412. doi:10.1093/her/cyp033.

Hamilton, A.J. *et al.* (2014) ‘Give peas a chance? Urban agriculture in developing countries. A review’, *Agronomy for Sustainable Development*, 34, pp. 45–73. doi:10.1007/s13593-013-0155-8.

Hamza, M.A. and Anderson, W.K. (2005) ‘Soil compaction in cropping systems: A review of the nature, causes and possible solutions’, *Soil and Tillage Research*, 82(2), pp. 121–145. doi:10.1016/j.still.2004.08.009.

Haney, R.L. *et al.* (2012) ‘Soil Organic C:N vs. Water-Extractable Organic C:N’, *Open Journal of Soil Science*, 2, pp. 269–274. doi:10.4236/ojss.2012.23032.

Hanssen, I. and Kuven, B.M. (2016) ‘Moments of joy and delight: the meaning of traditional food in dementia care’, *Journal of Clinical Nursing*, 25(5–6), pp. 866–874. doi:10.1111/jocn.13163.

Hardaker, A., Pagella, T. and Rayment, M. (2021) ‘Ecosystem service and dis-service impacts of increasing tree cover on agricultural land by land-sparing and land-sharing in the Welsh uplands’, *Ecosystem Services*, 48(101253). doi:10.1016/j.ecoser.2021.101253.

Harwood, J. (2011) *Europe’s Green Revolution and its Successors: The Rise and Fall of Peasant-Friendly Plant Breeding*. London, UK: Routledge. doi:10.4324/9780203118047.

Hassan, A., Dresbøll, D.B. and Thorup-Kristensen, K. (2021) ‘Naturally coloured roots as a tool for studying root interactions in mixed cropping’, *Plant, Soil and Environment*, 67(12), pp. 700–710. doi:10.17221/154/2021-PSE.

Hauschild, M. (2000) ‘Estimating pesticide emissions for LCA of agricultural products’, in Weidema, B. and Meeusen, M. (eds) *Agricultural Data for Life Cycle Assessments*. The Haugue: Agricultural Economics Research Institute, pp. 64–79.

Hauschild, M.Z. (2017) ‘Introduction to LCA methodology’, in Hauschild, M.Z., Rosenbaum, R.K., and Olsen, S.I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham, Switzerland: Springer International Publishing AG, pp. 59–66. doi:10.1007/978-3-319-56475-3_6.

Hauschild, M.Z. and Huijbregts, M.A.J. (2015) *Life Cycle Impact Assessment*. Dordrecht: Springer Netherlands (LCA Compendium – The Complete World of Life Cycle Assessment). doi:10.1007/978-94-017-9744-3.

Heath, S.K. *et al.* (2017) ‘A bustle in the hedgerow: Woody field margins boost on farm avian diversity and abundance in an intensive agricultural landscape’, *Biological Conservation*, 212, pp. 153–161. doi:10.1016/j.biocon.2017.05.031.

Hebinck, A. *et al.* (2021) ‘A Sustainability Compass for policy navigation to sustainable food systems’, *Global Food Security*, 29(100546). doi:10.1016/j.gfs.2021.100546.

Heflin, C.M., Corcoran, M.E. and Siefert, K.A. (2007) 'Work trajectories, income changes, and food insufficiency in a Michigan welfare population', *Social Service Review*, 81(1), pp. 3–25. doi:10.1086/511162.

Heijungs, R. *et al.* (1992) *Environmental life cycle assessment of products: guide and backgrounds (Part 1)*. Edited by R. Heijungs. Leiden, Netherlands: Centre of Environmental Science.

Heinrich, L.M. and Gullone, E. (2006) 'The clinical significance of loneliness: A literature review', *Clinical Psychology Review*, 26(6), pp. 695–718. doi:10.1016/j.cpr.2006.04.002.

Helalia, A.M. (1993) 'The relation between soil infiltration and effective porosity in different soils', *Agricultural Water Management*, 24(1), pp. 39–47. doi:10.1016/0378-3774(93)90060-N.

Helliwell, J.F. (2011) 'How Can Subjective Well-being Be Improved?', in Gorbet, F. and Sharpe, A. (eds) *New directions for intelligent government in Canada*. Ottawa, Canada: Centre for the Study of Living Standards, pp. 283–304.

Helliwell, J.F., Huang, H. and Wang, S. (2014) 'Social Capital and Well-Being in Times of Crisis', *Journal of Happiness Studies*, 15(1), pp. 145–162. doi:10.1007/s10902-013-9441-z.

Hendrickson, M.K. (2020) 'Covid lays bare the brittleness of a concentrated and consolidated food system', *Agriculture and Human Values*, 37(3), pp. 579–580. doi:10.1007/s10460-020-10092-y.

Henley, J. (2020) 'The future of food: inside the world's largest urban farm – built on a rooftop', *The Guardian*, 8 July. Available at: https://www.theguardian.com/cities/2020/jul/08/the-future-of-food-inside-the-worlds-largest-urban-farm-built-on-a-rooftop.

Henryson, K. *et al.* (2019) 'Environmental performance of crop cultivation at different sites and nitrogen rates in Sweden', *Nutrient Cycling in Agroecosystems*, 114(2), pp. 139–155. doi:10.1007/s10705-019-09997-w.

Henryson, K. *et al.* (2020) 'Soil N2O emissions, N leaching and marine eutrophication in life cycle assessment – A comparison of modelling approaches', *Science of the Total Environment*, 725(138332). doi:10.1016/j.scitotenv.2020.138332.

Hickey, G.M. and Unwin, N. (2020) 'Addressing the triple burden of malnutrition in the time of COVID-19 and climate change in Small Island Developing States: what role for improved local food production?', *Food Security*, 12, pp. 831–835. doi:10.1007/s12571-020-01066-3.

Hicks-Stratton, C. (2004) *The experience of food bank usage among women: a phenomenological study*. Memorial University of Newfoundland. Available at: http://research.library.mun.ca/id/eprint/10653.

Higashi, R.T. *et al.* (2017) 'Family and Social Context Contributes to the Interplay of Economic Insecurity, Food Insecurity, and Health', *Annals of Anthropological Practice*, 41(2), pp. 67–77. doi:10.1111/napa.12114.

Hilmers, A., Hilmers, D.C. and Dave, J. (2012) 'Neighborhood disparities in access to healthy foods and their effects on environmental justice', *American Journal of Public Health*, 102(9), pp. 1644–1654. doi:10.2105/AJPH.2012.300865.

HM Revenue & Customs (2022) *Self-Employment Income Support Scheme: understanding customer experience - qualitative research with customers and agents, Research and analysis*. Available at: https://www.gov.uk/government/publications/self-employed-income-support-scheme-understanding-customer-experience (Accessed: 17 October 2022).

Hobbs, J.E. (2020) 'Food supply chains during the COVID-19 pandemic', *Canadian Journal of Agricultural Economics*, 68(2), pp. 171–176. doi:10.1111/cjag.12237.

Hochmuth, G.J. and Hochmuth, R.C. (2018) *Nutrient Solution Formulation for Hydroponic (Perlite, Rockwool, NFT) Tomatoes in Florida, Soil and Water*. Gainesville, FL: UF/IFAS Extension Service, University of Florida. Available at: https://edis.ifas.ufl.edu/pdf/CV/CV21600.pdf.

Hodbod, J. and Eakin, H. (2015) ‘Adapting a social-ecological resilience framework for food systems’, *Journal of Environmental Studies and Sciences*, 5(3), pp. 474–484. doi:10.1007/s13412-015-0280-6.

Hodgson, J.A. *et al.* (2010) ‘Comparing organic farming and land sparing: optimizing yield and butterfly populations at a landscape scale’, *Ecology Letters*, 13(11), pp. 1358–1367. doi:10.1111/j.1461-0248.2010.01528.x.

Holcombe, M. (2019) ‘This southern city is fighting food deserts with a forest of free produce’, *CNN*, 24 May. Available at: https://edition.cnn.com/2019/05/24/us/atlanta-food-forest-fighting-food-desert/index.html.

Hole, D.G. *et al.* (2005) ‘Does organic farming benefit biodiversity?’, *Biological Conservation*, 122(1), pp. 113–130. doi:10.1016/j.biocon.2004.07.018.

Holling, C.S. (1973) ‘Resilience and Stability of Ecological Systems’, *Annual Review of Ecology and Systematics*, 4(1), pp. 1–23. doi:10.1146/annurev.es.04.110173.000245.

Holling, C.S. (1996) ‘Engineering Resilience versus Ecological Resilience’, in Schulze, P.E. (ed.) *Engineering within Ecological Constraints*. Washington DC, USA: National Academy Press, pp. 31–43.

Holman, B. (1998) *Faith in the poor*. Oxford, UK: Lion Publishing.

Holman, N. *et al.* (2020) ‘Risk factors for COVID-19-related mortality in people with type 1 and type 2 diabetes in England: a population-based cohort study’, *The Lancet Diabetes & Endocrinology*, 8(10), pp. 823–833. doi:10.1016/S2213-8587(20)30271-0.

Holmes, E.A. *et al.* (2020) ‘Multidisciplinary research priorities for the COVID-19 pandemic: a call for action for mental health science’, *The Lancet Psychiatry*, 7(6), pp. 547–560. doi:10.1016/S2215-0366(20)30168-1.

Holt-Giménez, E. *et al.* (2012) ‘We Already Grow Enough Food for 10 Billion People … and Still Can’t End Hunger’, *Journal of Sustainable Agriculture*, 36(6), pp. 595–598. doi:10.1080/10440046.2012.695331.

Holt-Giménez, E. and Altieri, M.A. (2013) ‘Agroecology, food sovereignty, and the new green revolution’, *Agroecology and Sustainable Food Systems*, 37(1), pp. 90–102. doi:10.1080/10440046.2012.716388.

van der Horst, H., Pascucci, S. and Bol, W. (2014) ‘The “dark side” of food banks? Exploring emotional responses of food bank receivers in the Netherlands’, *British Food Journal*, 116(9), pp. 1506–1520. doi:10.1108/BFJ-02-2014-0081.

Horton, P. *et al.* (2017) ‘An agenda for integrated system-wide interdisciplinary agri-food research’, *Food Security*, 9(2), pp. 195–210. doi:10.1007/s12571-017-0648-4.

Horton, P. (2017) ‘We need radical change in how we produce and consume food’, *Food Security*, 9(6), pp. 1323–1327. doi:10.1007/s12571-017-0740-9.

Horton, P., Koh, L. and Guang, V.S. (2016) ‘An integrated theoretical framework to enhance resource efficiency, sustainability and human health in agri-food systems’, *Journal of Cleaner Production*, 120, pp. 164–169. doi:10.1016/j.jclepro.2015.08.092.

Hosono, T. *et al.* (2006) ‘Measurements of N2O and NO emissions during tomato cultivation using a flow-through chamber system in a glasshouse’, *Nutrient Cycling in Agroecosystems*, 75(1–3), pp. 115–134. doi:10.1007/s10705-006-9016-z.

House, J.S. (2008) ‘Social psychology, social science, and economics: Twentieth century progress and problems, twenty-first century prospects', *Social Psychology Quarterly*, 71(3), pp. 232–256. doi:10.1177/019027250807100306.

HSE (2022) *HSE Pesticides eBulletin: Proposed GB Maximum Residue Levels – Chlorothalonil / Proposed Withdrawal of GB Approval of an Active Substance*, *HSE Pesticides eBulletin: Health and Safety Executive*. Available at: https://content.govdelivery.com/accounts/UKHSE/bulletins/312304b (Accessed: 15 December 2022).

Hsiang, S.M., Meng, K.C. and Cane, M.A. (2011) 'Civil conflicts are associated with the global climate', *Nature*, 476(7361), pp. 438–441. doi:10.1038/nature10311.

Hu, Y. *et al.* (2019) 'Carbon footprint and economic efficiency of urban agriculture in Beijing - a comparative case study of conventional and home-delivery agriculture', *Journal of Cleaner Production*, 234, pp. 615–625. doi:10.1016/j.jclepro.2019.06.122.

Hu, Y. *et al.* (2022) 'Fate of P from organic and inorganic fertilizers assessed by complementary approaches', *Nutrient Cycling in Agroecosystems*, 124(2), pp. 189–209. doi:10.1007/s10705-022-10237-x.

Huang, I.Y. *et al.* (2022) *Are English farmers ready for the changes in UK agricultural and environmental policy? Final Report*. Newport, UK: Harper Adams University. Available at: https://projectblue.blob.core.windows.net/media/Default/Horizon/Harper Adams SFI report April 22.pdf.

Hueso-Kortekaas, K., Romero, J.C. and González-Felipe, R. (2021) 'Energy-Environmental Impact Assessment of Greenhouse Grown Tomato: A Case Study in Almeria (Spain)', *World*, 2(3), pp. 425–441. doi:10.3390/world2030027.

Hughes, M.E. *et al.* (2004) 'A Short Scale for Measuring Loneliness in Large Surveys', *Research on Aging*, 26(6), pp. 655–672. doi:10.1177/0164027504268574.

Huijbregts, M.A.J. *et al.* (2017) 'ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level', *International Journal of Life Cycle Assessment*, 22(2), pp. 138–147. doi:10.1007/s11367-016-1246-y.

Hunter, P. (2008) 'A toxic brew we cannot live without', *EMBO reports*, 9(1), pp. 15–18. doi:10.1038/sj.embor.7401148.

Hwang, S.-J. (2020) 'Eutrophication and the Ecological Health Risk', *International Journal of Environmental Research and Public Health*, 17(17), p. 6332. doi:10.3390/ijerph17176332.

Hyder, O. *et al.* (2013) 'Cadmium Exposure and Liver Disease among US Adults', *Journal of Gastrointestinal Surgery*, 17(7), pp. 1265–1273. doi:10.1007/s11605-013-2210-9.

ICF Consulting (2005) *Determination of the Impact of Waste Management Activities on Greenhouse Gas Emissions: 2005 Update*. Toronto, Canada: Environment Canada and Natural Resources Canada. Available at: https://www.rcbc.ca/files/u3/ICF-final-report.pdf.

IFOAM (2019) *The IFOAM Norms for Organic Production and Processing Version 2014*. Bonn, Germany: International Federation of Organic Agriculture Movements. Available at: https://www.ifoam.bio/sites/default/files/2020-09/IFOAM Norms July 2014 Edits 2019.pdf.

IFOAM (2020) *Organic Agriculture & Biodiversity Factsheet*. Bonn, Germany: International Federation of Organic Agriculture Movements. Available at: https://www.ifoam.bio/sites/default/files/2020-03/oa_and_biodiversity_web.pdf.

Iob, E., Steptoe, A. and Fancourt, D. (2020) 'Abuse, self-harm and suicidal ideation in the UK during the COVID-19 pandemic', *The British Journal of Psychiatry*, 217(4), pp. 543–546. doi:10.1192/bjp.2020.130.

Ion, V. *et al.* (2014) 'Results regarding biomass yield at sunflower under different planting patterns and growing conditions', *Series A. Agronomy*, LVII, pp. 205–210.

IPBES (2018) *The IPBES assessment report on land degradation and restoration, Proceedings of the Global Symposium on Soil Organic Carbon 2017, Rome, Italy, 21-23 March, 2017*. Edited by M.R. Scholes et al. Bonn, Germany: Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services Secretariat, United Nations. Available at: https://ipbes.net/assessment-reports/ldr.

IPCC (2006a) 'Biological Treatment of Soild Waste', in Kruger, D. and Parikh, K. (eds) *IPCC Guidelines for National Greenhouse Gas Inventories: Volume 5, Waste*. Geneva, Switzerland: Institute for Global Environmental Strategies for the Intergovernmental Panel on Climate Change. Available at: https://www.ipcc-nggip.iges.or.jp/public/2006gl/vol5.html.

IPCC (2006b) 'Consistent representation of lands', in Eggelston, S. et al. (eds) *IPCC Guidelines for National Greenhouse Gas Inventories: Volume 4, Agriculture, Forestry, and Other Land Use*. Geneva, Switzerland: Institute for Global Environmental Strategies for the Intergovernmental Panel on Climate Change. Available at: https://www.ipcc.ch/report/2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/.

IPCC (2006c) 'N2O emissions from managed soils, and CO2 emissions from lime and urea application', in Eggelston, S. et al. (eds) *IPCC Guidelines for National Greenhouse Gas Inventories: Volume 4, Agriculture, Forestry, and Other Land Use*. Geneva, Switzerland: Institute for Global Environmental Strategies for the Intergovernmental Panel on Climate Change. Available at: https://www.ipcc.ch/report/2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/.

IPCC (2014) *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Edited by Core Writing Team, R.K. Pachauri, and L.A. Meyer. Geneva, Switzerland: Intergovernmental Panel on Climate Change (IPCC). Available at: https://www.ipcc.ch/report/ar5/syr/.

IPCC (2019) 'N2O emissions from managed soils, and CO2 emissions from lime and urea application', in Gómez, D. and Irving, W. (eds) *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*. Geneva, Switzerland: Institute for Global Environmental Strategies for the Intergovernmental Panel on Climate Change. Available at: https://www.ipcc.ch/report/2019-refinement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/.

IPCC (2022) 'Agriculture, Forestry and Other Land Uses (AFOLU)', in *Climate Change 2022: Mitigation of Climate Change. Working Group III Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Intergovernmental Panel on Climate Change. Available at: https://www.ipcc.ch/report/ar6/wg3/.

ISO (1995) *ISO 11466:1995 - Soil quality - Extraction of trace elements soluble in aqua regia*. Geneva, Switzerland: International Organization for Standardization. Available at: https://www.iso.org/standard/19418.html.

ISO (2006a) *ISO 14040:2006 Environmental management - Life cycle assessment - Principles and framework*. Geneva, Switzerland: International Organization for Standardization. doi:10.1016/j.ecolind.2011.01.007.

ISO (2006b) *ISO 14044:2006 Environmental management - Life cycle assessment - Requirements and guidelines*. Geneva, Switzerland: International Organization for Standardization. Available at: https://www.iso.org/standard/38498.html.

Ivosev, G., Burton, L. and Bonner, R. (2008) 'Dimensionality reduction and visualization in principal component analysis', *Analytical Chemistry*, 80(13), pp. 4933–4944. doi:10.1021/ac800110w.

Iwegbue, C.M.A. *et al.* (2007) 'Fractionation, characterization and speciation of heavy metals in composts and compost-amended soils', *African Journal of Biotechnology*, 6(2), pp. 067–078. doi:10.1002/chin.200748228.

Jackson, D.L. (2000) *Guidance on the interpretation of the Biodiversity Broad Habitat Classification (terrestrial and freshwater types): Definitions and the relationship with other classifications*. Available at: http://www.jncc.gov.uk/page-2433.

Jacob, D. *et al.* (2018) 'Climate Impacts in Europe Under +1.5°C Global Warming', *Earth's Future*, 6(2), pp. 264–285. doi:10.1002/2017EF000710.

Jaga, K. and Dharmani, C. (2005) 'The Epidemiology of Pesticide Exposure and Cancer: A Review', *Reviews on Environmental Health*, 20(1). doi:10.1515/REVEH.2005.20.1.15.

Jagtap, S. *et al.* (2022) 'The Russia-Ukraine Conflict: Its Implications for the Global Food Supply Chains', *Foods*, 11(14), pp. 1–23. doi:10.3390/foods11142098.

Jain, H.K. (2010) *The Green Revolution: history, impact and future*. Houston, TX, USA: Studium Press LLC.

Janssen, M. and Hamm, U. (2012) 'The mandatory EU logo for organic food: consumer perceptions', *British Food Journal*, 114(3), pp. 335–352. doi:10.1108/00070701211213456.

Jarmul, S. *et al.* (2020) 'Climate change mitigation through dietary change: a systematic review of empirical and modelling studies on the environmental footprints and health effects of "sustainable diets"', *Environmental Research Letters*, 15(12), p. 123014. doi:10.1088/1748-9326/abc2f7.

Jarvis, S.C. *et al.* (1996) 'Nitrogen mineralization in temperate agricultural soils: processes and measurement', in Sparks, D.L. (ed.) *Advances in Agronomy (Vol 57)*. Academic Press Inc., Elsevier Science, pp. 187–235.

Jeanneret, P. *et al.* (2014) 'An expert system for integrating biodiversity into agricultural life-cycle assessment', *Ecological Indicators*, 46, pp. 224–231. doi:10.1016/j.ecolind.2014.06.030.

Jeffery, S. and Verheijen, F.G.A. (2020) 'A New soil health policy paradigm: Pay for practice not performance!', *Environmental Science and Policy*, 112, pp. 371–373. doi:10.1016/j.envsci.2020.07.006.

Jenkins, R.H. *et al.* (2021) 'The relationship between austerity and food insecurity in the UK: A systematic review', *EClinicalMedicine*, 33(100781). doi:10.1016/j.eclinm.2021.100781.

Jetter, K.M. and Cassady, D.L. (2006) 'The Availability and Cost of Healthier Food Alternatives', *American Journal of Preventive Medicine*, 30(1), pp. 38–44. doi:10.1016/j.amepre.2005.08.039.

John, D.A. and Babu, G.R. (2021) 'Lessons From the Aftermaths of Green Revolution on Food System and Health', *Frontiers in Sustainable Food Systems*, 5. doi:10.3389/fsufs.2021.644559.

Johnston, A.E. (1975) 'The Woburn Market Garden experiment, 1942–69. II. Effects of the treatments on soil pH, soil carbon, nitrogen, phosphorus and potassium.', *Rothamsted Experimental Station Report*, 2, pp. 102–132. doi:10.23637/ERADOC-1-33161.

Johnston, A.E., Poulton, P.R. and Coleman, K. (2009) 'Soil Organic Matter: Its Importance in Sustainable Agriculture and Carbon Dioxide Fluxes', in Sparks, D.L. (ed.) *Advances in Agronomy (Vol 101)*. Academic Press, pp. 1–57. doi:10.1016/S0065-2113(08)00801-8.

Johnston, A.M. and Bruulsema, T.W. (2014) '4R Nutrient Stewardship for Improved Nutrient Use Efficiency', *Procedia Engineering*, 83, pp. 365–370. doi:10.1016/j.proeng.2014.09.029.

Jones, M. *et al.* (2020) 'Apart but not Alone? A cross-sectional study of neighbour support in a major UK urban area during the COVID-19 lockdown', *Emerald Open Research*, 2, p. 37. doi:10.35241/emeraldopenres.13731.1.

Jones, S., Krzywoszynska, A. and Maye, Damian (2022) 'Resilience and transformation: Lessons from the UK local food sector in the COVID-19 pandemic', *Geographical Journal*, 188(2), pp. 209–222. doi:10.1111/geoj.12428.

Jones, S., Krzywoszynska, A. and Maye, D. (2022) *Resilience and transformation. Resilience of the UK's local food sector to the first wave of the COVID-19 pandemic*. doi:10.5281/zenodo.6380015.

Joseph, J. (2013) 'Resilience as embedded neoliberalism: a governmentality approach', *Resilience*, 1(1), pp. 38–52. doi:10.1080/21693293.2013.765741.

Justes, E., Mary, B. and Nicolardot, B. (2009) 'Quantifying and modelling C and N mineralization kinetics of catch crop residues in soil: Parameterization of the residue decomposition module of STICS model for mature and non mature residues', *Plant and Soil*, 325(1), pp. 171–185. doi:10.1007/s11104-009-9966-4.

Kafle, A., Hopeward, J. and Myers, B. (2022) 'Exploring Conventional Economic Viability as a Potential Barrier to Scalable Urban Agriculture: Examples from Two Divergent Development Contexts', *Horticulturae*, 8(8), p. 691. doi:10.3390/horticulturae8080691.

Kamilaris, A. and Prenafeta-Boldú, F.X. (2021) 'Examining the perspectives of using manure from livestock farms as fertilizer to crop fields based on a realistic simulation', *Computers and Electronics in Agriculture*, 191(106486). doi:10.1016/j.compag.2021.106486.

Karlen, D.L. *et al.* (1997) 'Soil Quality: A Concept, Definition, and Framework for Evaluation (A Guest Editorial)', *Soil Science Society of America Journal*, 61(1), pp. 4–10. doi:10.2136/sssaj1997.03615995006100010001x.

Karlen, D.L. (2012) 'Soil Health: The Concept, Its Role, and Strategies for Monitoring', in Wall, D.H. et al. (eds) *Soil Ecology and Ecosystem Services*. Oxford, UK: Oxford University Press, pp. 331–336. doi:10.1093/acprof:oso/9780199575923.003.0029.

Karlowsky, S. *et al.* (2021) 'Seasonal Nitrous Oxide Emissions From Hydroponic Tomato and Cucumber Cultivation in a Commercial Greenhouse Company', *Frontiers in Sustainable Food Systems*, 5(626053). doi:10.3389/fsufs.2021.626053.

Karlsson, H. (2011) *Seasonal Vegetables: An Environmental Assessment Seasonal Food*. Norwegian University of Life Sciences. Available at: https://brage.bibsys.no/xmlui/bitstream/handle/11250/189423/Hanna Karlsson.pdf?sequence=1.

Karlsson, J.O., Tidåker, P. and Röös, E. (2022) 'Smaller farm size and ruminant animals are associated with increased supply of non-provisioning ecosystem services', *Ambio*, 51(9), pp. 2025–2042. doi:10.1007/s13280-022-01726-y.

Kavada, A. (2022) 'Creating a Hyperlocal Infrastructure of Care: COVID-19 Mutual Aid Groups in the UK', *Social Movements and Politics During COVID-19*, pp. 147–154. doi:10.51952/9781529217254.ch019.

Keane, M. and Neal, T. (2021) 'Consumer panic in the COVID-19 pandemic', *Journal of Econometrics*, 220(1), pp. 86–105. doi:10.1016/j.jeconom.2020.07.045.

Keesstra, S. *et al.* (2018) 'The superior effect of nature based solutions in land management for enhancing ecosystem services', *Science of The Total Environment*, 610–611, pp. 997–1009. doi:10.1016/j.scitotenv.2017.08.077.

Keller, M. *et al.* (2012) 'Phosphorus forms and enzymatic hydrolyzability of organic phosphorus in soils after 30 years of organic and conventional farming', *Journal of Plant Nutrition and Soil Science*, 175(3), pp. 385–393. doi:10.1002/jpln.201100177.

Kemble, J.M. (2019) *Southeastern U.S. 2019 Vegetable Crop HandBook*. Willoughby, OH, USA: Southeastern Vegetable Extension Workers Group. Available at: https://pubs.ext.vt.edu/AREC/AREC-66/AREC-66.html.

Kemp, K. *et al.* (2010) 'Food miles: Do UK consumers actually care?', *Food Policy*, 35(6), pp. 504–513. doi:10.1016/j.foodpol.2010.05.011.

Kennard, N.J. (2019) ‘Food Waste Management’, in Leal Filho, W. et al. (eds) *Zero Hunger. Encyclopedia of the UN Sustainable Development Goals*. Cham: Springer. doi:https://doi.org/10.1007/978-3-319-69626-3.

Kennard, N.J. (2023) *Experiences accessing food during COVID-19: A Case Study with Foodhall, Sheffield*. Report. University of Sheffield. doi:10.15131/shef.data.23694957.v2.

Kennard, N.J. and Bamford, R.H. (2020) ‘Urban Agriculture: Opportunities and Challenges for Sustainable Development’, in Leal Filho, W. et al. (eds) *Zero Hunger. Encyclopedia of the UN Sustainable Development Goals*. Cham: Springer, pp. 929–942. doi:10.1007/978-3-319-95675-6_102.

Kennard, N.J. and Vagnoni, C. (2021) *Pesticides and health: POSTbrief 43*. London, UK: UK Parliamentary Office of Science & Technology. Available at: https://post.parliament.uk/research-briefings/post-pb-0043/.

Khan, N. (2022) ‘The cost of living crisis: how can we tackle fuel poverty and food insecurity in practice?’, *British Journal of General Practice*, 72(720), pp. 330–331. doi:10.3399/bjgp22X719921.

Killpack, S.C. and Buchholz, D. (1993) *Nitrogen in the Environment: Leaching, University of Missouri Extension*. Available at: https://extension.missouri.edu/publications/wq262 (Accessed: 27 August 2022).

Kirchmann, H. *et al.* (2009) ‘Can Organic Crop Production Feed the World?’, in Kirchmann, H. and Bergström, L. (eds) *Organic Crop Production – Ambitions and Limitations*. Dordrecht, Netherlands: Springer, pp. 39–72. doi:10.1007/978-1-4020-9316-6_3.

Kirchmann, H. and Bergqvist, R. (1989) ‘Carbon and nitrogen mineralization of white clover plants (Trifolium repens) of different age during aerobic incubation with soil’, *Zeitschrift für Pflanzenernährung und Bodenkunde*, 152(3), pp. 281–286. doi:10.1002/jpln.19891520303.

Kirchmann, H. and Bergström, L. (2001) ‘Do organic farming practices reduce nitrate leaching?’, *Communications in Soil Science and Plant Analysis*, 32(7–8), pp. 997–1028. doi:10.1081/CSS-100104101.

Klassen, S.E. and Wittman, H. (2017) ‘Place-based food systems’, in Duncan, J. and Bailey, M. (eds) *Sustainable Food Futures: Multidiscplinary Solutions*. 1st edn. London, UK: Routledge. doi:10.4324/9781315463131.

Kleijn, D. *et al.* (2009) ‘On the relationship between farmland biodiversity and land-use intensity in Europe’, *Proceedings of the Royal Society B: Biological Sciences*, 276(1658), pp. 903–909. doi:10.1098/rspb.2008.1509.

Kleinman, P.J.A. *et al.* (2002) ‘Effect of Mineral and Manure Phosphorus Sources on Runoff Phosphorus’, *Journal of Environmental Quality*, 31(6), pp. 2026–2033. doi:10.2134/jeq2002.2026.

Kleinman, P.J.A. *et al.* (2011) ‘Managing agricultural phosphorus for water quality protection: principles for progress’, *Plant and Soil*, 349(1–2), pp. 169–182. doi:10.1007/s11104-011-0832-9.

Klinenberg, E. (2003) *Heat wave: A social autopsy of disaster in Chicago*. Chicago, IL: University of Chicago Press.

Kneafsey, M. *et al.* (2013) *Short food supply chains and local food systems in the EU (EUR 25911 EN; JRC Scientific and Policy Reports)*. Edited by F. Santini and S. Gomez y Paloma. Luxembourg: Publications Office of the European Union. doi:10.2791/88784.

Koch, P. and Thibault, S. (2020) *AGRIBALYSE: Methodology, Agricultural stage - Version 3.0*. Angers, France: ADEME.

Kolodinksy, J. and Cranwell, M. (2000) ‘The Poor Pay More? Now They Don’t Even Have a Store to Choose From: Bringing a Supermarket Back to the City’, *Consumer Interests Annual*, 46, pp. 24–29.

Komilis, D.P. and Ham, R.K. (2004) ‘Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States’, *Journal of Environmental Engineering*, 130(11), pp. 1390–1400. doi:10.1061/(asce)0733-9372(2004)130:11(1390).

Komilis, D.P., Ham, R.K. and Park, J.K. (2004) ‘Emission of volatile organic compounds during composting of municipal solid wastes’, *Water Research*, 38(7), pp. 1707–1714. doi:10.1016/j.watres.2003.12.039.

Kong, D. *et al.* (2012) ‘Evaluating greenhouse gas impacts of organic waste management options using life cycle assessment’, *Waste Management and Research*, 30(8), pp. 800–812. doi:10.1177/0734242X12440479.

Koont, S. (2008) ‘A Cuban Success Story: Urban Agriculture’, *Review of Radical Political Economics*, 40(3), pp. 285–291. doi:10.1177/0486613408320016.

Korus, A. (2020) ‘Changes in the content of minerals, B-group vitamins and tocopherols in processed kale leaves’, *Journal of Food Composition and Analysis*, 89(103464). doi:10.1016/j.jfca.2020.103464.

Koseda, L. (2020) *Covid has shown why we always need social architeture*, *The RIBA Journal*. Available at: https://www.ribaj.com/culture/coronavirus-national-food-service-social-architecture-sheffield-foodhall.

Kosheleva, N.E. and Nikiforova, E.M. (2016) ‘Long-Term Dynamics of Urban Soil Pollution with Heavy Metals in Moscow’, *Applied and Environmental Soil Science*, pp. 1–10. doi:10.1155/2016/5602795.

Koswatta, T.J. *et al.* (2023) ‘Factors affecting public perception of scientific information about organic foods’, *British Food Journal*, 125(2), pp. 587–607. doi:10.1108/BFJ-08-2021-0874.

Kousemaker, T.M., Jonker, G.H. and Vakis, A.I. (2021) ‘LCA practices of plastics and their recycling: A critical review’, *Applied Sciences (Switzerland)*, 11(8), pp. 1–17. doi:10.3390/app11083305.

Koutroulis, A.G. (2019) ‘Dryland changes under different levels of global warming’, *Science of The Total Environment*, 655, pp. 482–511. doi:10.1016/j.scitotenv.2018.11.215.

Kranz, C.N. *et al.* (2020) ‘The effects of compost incorporation on soil physical properties in urban soils – A concise review’, *Journal of Environmental Management*, 261, p. 110209. doi:10.1016/j.jenvman.2020.110209.

Kremen, C., Iles, A. and Bacon, C. (2012) ‘Diversified farming systems: An agroecological, systems-based alternative to modern industrial agriculture’, *Ecology and Society*, 17(4). doi:10.5751/ES-05103-170444.

Kremen, C. and Miles, A. (2012) ‘Ecosystem services in biologically diversified versus conventional farming systems: Benefits, externalities, and trade-offs’, *Ecology and Society*, 17(4). doi:10.5751/ES-05035-170440.

Kroger (2022) *Nurturing Shared Values: Kroger ESG Report 2022*. Available at: https://www.thekrogerco.com/wp-content/uploads/2022/08/Kroger-Co-2022-ESG-Report.pdf.

Kropotkin, P. (1902) *Mutual Aid: A Factor in Evolution*. New York, NY: McClure, Philips, & Company.

Krzywoszynska, A., Jones, S. and Maye, D. (2022) *Riding the waves. The long perspective on the COVID-19 pandemic from UK’s local food system actors in 2020-21*. doi:10.5281/zenodo.6380040.

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

Kumar, K. and Hundal, L.S. (2016) ‘Soil in the City: Sustainably Improving Urban Soils’, *Journal of Environment Quality*, 45(1), p. 2. doi:10.2134/jeq2015.11.0589.

Kumar, R. *et al.* (2014) ‘Green manuring: A boon for sustainable agriculture and pest management – A review’, *Agricultural Reviews*, 35(3), p. 196. doi:10.5958/0976-0741.2014.00906.4.

Kuntoji, A. *et al.* (2021) ‘Nutrient content of tomato as influenced by different sources and levels of boron’, *Journal of the Indian Society of Soil Science*, 69(2), pp. 203–209. doi:10.5958/0974-0228.2021.00028.1.

Kwong, A.S.F. *et al.* (2021) ‘Mental health before and during the COVID-19 pandemic in two longitudinal UK population cohorts’, *The British Journal of Psychiatry*, 218(6), pp. 334–343. doi:10.1192/bjp.2020.242.

Laborde, D. *et al.* (2020) ‘COVID-19 risks to global food security’, *Science*, 369(6503), pp. 500–502. doi:10.1126/science.abc4765.

Lal, R. (2004) ‘Soil carbon sequestration to mitigate climate change’, *Geoderma*, 123(1–2), pp. 1–22. doi:10.1016/j.geoderma.2004.01.032.

Lal, R. (2010) ‘Beyond Copenhagen: Mitigating climate change and achieving food security through soil carbon sequestration’, *Food Security*, 2, pp. 169–177. doi:10.1007/s12571-010-0060-9.

Lal, R. (2014) ‘Societal value of soil carbon’, *Journal of Soil and Water Conservation*, 69(6), pp. 186A-192A. doi:10.2489/jswc.69.6.186A.

Lal, R. (2015) ‘Restoring soil quality to mitigate soil degradation’, *Sustainability*, 7(5), pp. 5875–5895. doi:10.3390/su7055875.

Lal, R. (2020) ‘Home gardening and urban agriculture for advancing food and nutritional security in response to the COVID-19 pandemic’, *Food Security*, pp. 871–876. doi:10.1007/s12571-020-01058-3.

Lambie-Mumford, H., Loopstra, R. and Gordon, K. (2020) ‘Mapping responses to risk of rising food insecurity during the COVID-19 crisis across the UK’. Available at: http://speri.dept.shef.ac.uk/food-vulnerability-during-covid-19/.

Lamichhane, J.R. *et al.* (2015) ‘Robust cropping systems to tackle pests under climate change. A review’, *Agronomy for Sustainable Development*, 35(2), pp. 443–459. doi:10.1007/s13593-014-0275-9.

Lang, G. and Miao, B. (2013) ‘Food Security for China’s Cities’, *International Planning Studies*, 18(1), pp. 5–20. doi:10.1080/13563475.2013.750940.

Lang, T. (2010) ‘Crisis? What crisis? The normality of the current food crisis’, *Journal of Agrarian Change*, 10(1), pp. 87–97. doi:10.1111/j.1471-0366.2009.00250.x.

Langemeyer, J. *et al.* (2021) ‘Urban agriculture — A necessary pathway towards urban resilience and global sustainability?’, *Landscape and Urban Planning*, 210(104055). doi:10.1016/j.landurbplan.2021.104055.

Langenhoven, P. (2018) *Hydroponic Tomato Production in Soilless Culture*. West Lafayette, IN, USA: Purdue University Extension. Available at: https://sp2013.ag.itap.purdue.edu/hla/fruitveg/Pages/IHC2018.aspx.

Łapiński, D. and Wiater, J. (2019) ‘Comparison of the Chemical Composition of Natural Fertilizers and Organic Waste’, *Proceedings*, 16(45). doi:10.3390/proceedings2019016045.

Lashermes, G. *et al.* (2022) ‘N2O emissions from decomposing crop residues are strongly linked to their initial soluble fraction and early C mineralization’, *Science of the Total Environment*, 806. doi:10.1016/j.scitotenv.2021.150883.

Laughton, R. (2017) *A Matter of Scale: A study of the productivity, financial viability and multifunctional benefits of small farms (20 ha and less)*. UK. Available at: https://landworkersalliance.org.uk/wp-content/uploads/2018/10/matterofscale.pdf.

Laurent, A., Espinosa, N. and Hauschild, M.Z. (2017) 'LCA of Energy Systems', in Hauschild, M.Z., Rosenbaum, R.K., and Olsen, S.I. (eds) *Life Cycle Assessment: Theory and Practice*. Cham, Switzerland: Springer International Publishing AG, pp. 633–668.

Lawley, R. (2009) *The Soil-Parent Material Database (SPM-v4): A User Guide*. Nottingham, UK. Available at: http://nora.nerc.ac.uk/8048/.

Lefsrud, M. *et al.* (2008) 'Dry matter content and stability of carotenoids in kale and spinach during drying', *HortScience*, 43(6), pp. 1731–1736. doi:10.21273/hortsci.43.6.1731.

Lehman, R.M. *et al.* (2015) 'Soil biology for resilient, healthy soil', *Journal of Soil and Water Conservation*, 70(1), pp. 12A-18A. doi:10.2489/jswc.70.1.12A.

Lehmann, J. *et al.* (2020) 'The concept and future prospects of soil health', *Nature Reviews Earth and Environment*, 1(10), pp. 544–553. doi:10.1038/s43017-020-0080-8.

Leifeld, J. *et al.* (2013) 'Organic farming gives no climate change benefit through soil carbon sequestration', *Proceedings of the National Academy of Sciences*, 110(11). doi:10.1073/pnas.1220724110.

Leifeld, J. and Fuhrer, J. (2010) 'Organic farming and soil carbon sequestration: What do we really know about the benefits?', *Ambio*, 39(8), pp. 585–599. doi:10.1007/s13280-010-0082-8.

Lemming, C. *et al.* (2019) 'Residual phosphorus availability after long-term soil application of organic waste', *Agriculture, Ecosystems and Environment*, 270–271, pp. 65–75. doi:10.1016/j.agee.2018.10.009.

Lepp, N.W. (1981) *Effect of Heavy Metal Pollution on Plants*. 1st edn. Dordrecht: Springer. doi:10.1007/978-94-011-7339-1.

Lesschen, J.P. *et al.* (2007) 'A spatially explicit methodology to quantify soil nutrient balances and their uncertainties at the national level', *Nutrient Cycling in Agroecosystems*, 78(2), pp. 111–131. doi:10.1007/s10705-006-9078-y.

Lever, J., Sonnino, R. and Cheetham, F. (2019) 'Reconfiguring local food governance in an age of austerity: towards a place-based approach?', *Journal of Rural Studies*, 69, pp. 97–105. doi:10.1016/j.jrurstud.2019.04.009.

Li, G. (2015) 'Investigations of life cycle climate performance and material life cycle assessment of packaged air conditioners for residential application', *Sustainable Energy Technologies and Assessments*, 11, pp. 114–125. doi:10.1016/j.seta.2015.07.002.

Li, G. *et al.* (2018) 'Urban soil and human health: a review', *European Journal of Soil Science*, 69(1), pp. 196–215. doi:10.1111/ejss.12518.

Li, L.Z. and Wang, S. (2020) 'Prevalence and predictors of general psychiatric disorders and loneliness during COVID-19 in the United Kingdom', *Psychiatry Research*, 291(113267). doi:10.1016/j.psychres.2020.113267.

Li, M. *et al.* (2022) 'Global food-miles account for nearly 20% of total food-systems emissions', *Nature Food*, 3(6), pp. 445–453. doi:10.1038/s43016-022-00531-w.

Li, Y. (2007) 'Social capital, social exclusion and wellbeing', in Scriven, A. and Garman, S. (eds) *Public Health: Social context and action*. Sage Publications Ltd, pp. 60–75.

Li, Y. (2015) 'Social capital in sociological research: conceptual rigour and empirical application', in Li, Y. (ed.) *The handbook of research methods and applications on social capital*. Cheltenham, UK:

Edward Elgar Publishing, pp. 1–20.

Li, Y., Pickles, A. and Savage, M. (2005) ‘Social capital and social trust in Britain’, *European Sociological Review*, 21(2), pp. 109–123. doi:10.1093/esr/jci007.

Libohova, Z. *et al.* (2018) ‘Reevaluating the effects of soil organic matter and other properties on available water-holding capacity using the National Cooperative Soil Survey Characterization Database’, *Journal of Soil and Water Conservation*, 73(4), pp. 411–421. doi:10.2489/jswc.73.4.411.

Liebert, J. *et al.* (2022) ‘Farm size affects the use of agroecological practices on organic farms in the United States’, *Nature Plants*, 8(8), pp. 897–905. doi:10.1038/s41477-022-01191-1.

Lin, B.B. and Fuller, R.A. (2013) ‘Sharing or sparing? How should we grow the world’s cities?’, *Journal of Applied Ecology*, 50, pp. 1161–1168. doi:10.1111/1365-2664.12118.

Lin, Brenda B, Philpott, S.M. and Jha, S. (2015) ‘The future of urban agriculture and biodiversity-ecosystem services: Challenges and next steps’, *Basic and Applied Ecology*, 16(3), pp. 189–201. doi:10.1016/j.baae.2015.01.005.

Lin, Brenda B., Philpott, S.M. and Jha, S. (2015) ‘The future of urban agriculture and biodiversity-ecosystem services: Challenges and next steps’, *Basic and Applied Ecology*, 16(3), pp. 189–201. doi:10.1016/j.baae.2015.01.005.

Lin, H., Lloyd, T. and McCorriston, S. (2021) ‘An Odd Crisis: Covid-19 and UK Food Prices’, *EuroChoices*, 19(3), pp. 42–48. doi:10.1111/1746-692X.12291.

Lin, N. (2001) *Social capital*. Cambridge, UK: Cambridge University Press.

Lindenthal, T. *et al.* (2010) ‘Greenhouse gas emissions of organic and conventional foodstuffs in Austria’, in *VII. International conference on life cycle assessment in the agri-food sector*. Bari, Italy. Available at: https://orgprints.org/id/eprint/17996/.

Linzner, R. and Mostbauer, P. (2005) ‘Composting and Its Impact on Climate Change With Regard To Process Engineering and Compost Application - a Case Study in Vienna’, in *Proceedings of Sardinia 2005, Tenth International Landfill Symposium*. Cagliari, Italy: CISA Publisher.

Liu, J. *et al.* (2010) ‘A high-resolution assessment on global nitrogen flows in cropland’, *Proceedings of the National Academy of Sciences of the United States of America*, 107(17), pp. 8035–8040. doi:10.1073/pnas.0913658107.

Liu, X. *et al.* (2006) ‘Effects of agricultural management on soil organic matter and carbon transformation - A review’, *Plant, Soil and Environment*, 52(12), pp. 531–543. doi:10.17221/3544-pse.

Llanos, J. and Border, P. (2020) *A resilient UK food system: POSTnote 626*. London, UK: UK Parliamentary Office of Science and Technology.

Llorach-Massana, P. *et al.* (2017) ‘N2O emissions from protected soilless crops for more precise food and urban agriculture life cycle assessments’, *Journal of Cleaner Production*, 149, pp. 1118–1126. doi:10.1016/j.jclepro.2017.02.191.

Lo, P.L., Wallis, R. and Bellamy, D.E. (2019) ‘The effectiveness of two types of adhesive for catching insects in traps’, *New Zealand Plant Protection*, 72, pp. 230–236. doi:10.30843/nzpp.2019.72.301.

Logsdon, S.D., Sauer, P.A. and Shipitalo, M.J. (2017) ‘Compost Improves Urban Soil and Water Quality’, *Journal of Water Resource and Protection*, 9(4), pp. 345–357. doi:10.4236/jwarp.2017.94023.

Long, E. *et al.* (2022) ‘COVID-19 pandemic and its impact on social relationships and health’, *Journal of epidemiology and community health*, 76(2), pp. 128–132. doi:10.1136/jech-2021-216690.

Long, M.A. *et al.* (2020) ‘Food Insecurity in Advanced Capitalist Nations: A Review’, *Sustainability*, 12(9), p. 3654. doi:10.3390/su12093654.

Loopstra, R. *et al.* (2019) *A survey of food banks operating independently of The Trussell Trust food bank network*. Available at: http://www.foodaidnetwork.org.uk/.

Loopstra, R. (2020) *Vulnerability to food insecurity since the COVID-19 lockdown: Preliminary report*. London, UK: Food Foundation. Available at: https://foodfoundation.org.uk/wp-content/uploads/2020/04/Report_COVID19FoodInsecurity-final.pdf.

Loopstra, R., Reeves, A. and Lambie-Mumford, H. (2020) *COVID-19: What impacts are unemployment and the Coronavirus Job Retention scheme having on food insecurity in the UK?* London, UK: The Food Foundation. Available at: https://foodfoundation.org.uk/sites/default/files/2021-09/BriefReport_Unemployment_v5.pdf.

Loopstra, R., Reeves, A. and Tarasuk, V. (2019) ‘The rise of hunger among low-income households: An analysis of the risks of food insecurity between 2004 and 2016 in a population-based study of UK adults’, *Journal of Epidemiology and Community Health*, 73(7), pp. 668–673. doi:10.1136/jech-2018-211194.

Loopstra, R. and Tarasuk, V. (2012) ‘The relationship between food banks and household food insecurity among low-income Toronto Families’, *Canadian Public Policy*, 38(4), pp. 497–514. doi:10.3138/CPP.38.4.497.

Lorenz, K. and Lal, R. (2005) ‘The Depth Distribution of Soil Organic Carbon in Relation to Land Use and Management and the Potential of Carbon Sequestration in Subsoil’, *Advances in Agronomy*, 88, pp. 35–66.

Lorenz, K. and Lal, R. (2016) ‘Environmental Impact of Organic Agriculture’, in Sparks, D.L. (ed.) *Advances in Agronomy*, pp. 99–152. doi:10.1016/bs.agron.2016.05.003.

Lotter, D.W. (2003) ‘Organic Agriculture’, *Journal of Sustainable Agriculture*, 21(4), pp. 59–128. doi:10.1300/J064v21n04_06.

Lovell, S.T. (2010) ‘Multifunctional Urban Agriculture for Sustainable Land Use Planning in the United States’, *Sustainability*, 2(8), pp. 2499–2522. doi:10.3390/su2082499.

Lovell, S.T. and Taylor, J.R. (2013) ‘Supplying urban ecosystem services through multifunctional green infrastructure in the United States’, *Landscape Ecology*, 28, pp. 1447–1463. doi:10.1007/s10980-013-9912-y.

De Luca, A.I. *et al.* (2015) ‘Social life cycle assessment and participatory approaches: A methodological proposal applied to citrus farming in Southern Italy’, *Integrated Environmental Assessment and Management*, 11(3), pp. 383–396. doi:10.1002/ieam.1611.

Lucas, R.E. and Davis, J.F. (1961) ‘Relationships between pH values of organic soils and availabilities of 12 plant nutrients’, *Soil Science*, 92(3), pp. 177–182. doi:10.1097/00010694-196109000-00005.

Lund, C. *et al.* (2011) ‘Poverty and mental disorders: Breaking the cycle in low-income and middle-income countries’, *The Lancet*, 378(9801), pp. 1502–1514. doi:10.1016/S0140-6736(11)60754-X.

Luo, L. and Wildemuth, B.M. (2017) ‘Semistructured Interviews’, in Wildemuth, B.M. (ed.) *Applications of social research methods to questions in information and library science*. 2nd edn. Santa Barbara, CA, USA: Libraries Unlimited, pp. 248–257.

Lutwak, N. and Ferrari, J.R. (1997) ‘Shame-related social anxiety: Replicating a link with various social interaction measures’, *Anxiety, Stress, & Coping*, 10(4), pp. 335–340. doi:10.1080/10615809708249307.

Lyons, K. and Westoby, P. (2014) ‘Carbon colonialism and the new land grab: Plantation forestry in

Uganda and its livelihood impacts', *Journal of Rural Studies*, 36, pp. 13–21. doi:10.1016/j.jrurstud.2014.06.002.

Maaoui, M., Boukchina, R. and Hajjaji, N. (2021) 'Environmental life cycle assessment of Mediterranean tomato: case study of a Tunisian soilless geothermal multi-tunnel greenhouse', *Environment, Development and Sustainability*, 23(2), pp. 1242–1263. doi:10.1007/s10668-020-00618-z.

Machado, R.M.A. and Oliveira, M.D.R.G. (2005) 'Tomato root distribution, yield and fruit quality under different subsurface drip irrigation regimes and depths', *Irrigation Science*, 24(1), pp. 15–24. doi:10.1007/s00271-005-0002-z.

Mackley, A. (2021) *Coronavirus : Universal Credit during the crisis*. London, UK: House of Commons Library. Available at: https://commonslibrary.parliament.uk/research-briefings/cbp-8999/.

Maeder, P. *et al.* (2002) 'Soil Fertility and Biodiversity in Organic Farming', *Science*, 296(5573), pp. 1694–1697. doi:10.1126/science.1071148.

Magdoff, F. and Weil, R.R. (2004) 'Soil Organic Matter Management Strategies', in Magdoff, F. and Weil, R.R. (eds) *Soil Organic Matter in Sustainable Agriculture*. Boca Raton, FL, USA: CRC Press, pp. 45–66.

Maher, M.J. (1976) 'Growth and nutrient content of a glasshouse tomato crop grown in peat', *Scientia Horticulturae*, 4(1), pp. 23–26. doi:10.1016/0304-4238(76)90060-1.

Maiter, S. *et al.* (2008) 'Reciprocity: An ethic for community-based participatory action research', *Action Research*, 6(3), pp. 305–325. doi:10.1177/1476750307083720.

Mammen, S., Bauer, J.W. and Richards, L. (2009) 'Understanding Persistent Food Insecurity: A Paradox of Place and Circumstance', *Social Indicators Research*, 92(1), pp. 151–168. doi:10.1007/s11205-008-9294-8.

Mangla, S.K. *et al.* (2018) 'Enablers to implement sustainable initiatives in agri-food supply chains', *International Journal of Production Economics*, 203, pp. 379–393. doi:10.1016/j.ijpe.2018.07.012.

Maniadakis, K. *et al.* (2004) 'Integrated Waste Management Through Producers and Consumers Education: Composting of Vegetable Crop Residues for Reuse in Cultivation', *Journal of Environmental Science and Health - Part B Pesticides, Food Contaminants, and Agricultural Wastes*, 39(1), pp. 169–183. doi:10.1081/PFC-120027447.

Manlay, R.J., Feller, C. and Swift, M.J. (2007) 'Historical evolution of soil organic matter concepts and their relationships with the fertility and sustainability of cropping systems', *Agriculture, Ecosystems & Environment*, 119(3–4), pp. 217–233. doi:10.1016/j.agee.2006.07.011.

Mann, C. *et al.* (2019) 'Relationships between field management, soil health, and microbial community composition', *Applied Soil Ecology*, 144, pp. 12–21. doi:10.1016/j.apsoil.2019.06.012.

Mao, G., Drury, J., *et al.* (2021) 'How participation in Covid-19 mutual aid groups affects subjective well-being and how political identity moderates these effects', *Analyses of Social Issues and Public Policy*, 21(1), pp. 1082–1112. doi:10.1111/asap.12275.

Mao, G., Fernandes-Jesus, M., *et al.* (2021) 'What have we learned about COVID-19 volunteering in the UK? A rapid review of the literature', *BMC Public Health*, 21(1). doi:10.1186/s12889-021-11390-8.

Mapp, T. (2008) 'Understanding phenomenology: the lived experience', *British Journal of Midwifery*, 16(5), pp. 308–311. doi:10.12968/bjom.2008.16.5.29192.

Di Marcantonio, F., Twum, E.K. and Russo, C. (2021) 'COVID-19 pandemic and food waste: An empirical analysis', *Agronomy*, 11(6), pp. 1–14. doi:10.3390/agronomy11061063.

De Marco, M., Thorburn, S. and Kue, J. (2009) '"In a country as affluent as America, people should be eating": Experiences with and perceptions of food insecurity among rural and urban oregonians', *Qualitative Health Research*, 19(7), pp. 1010–1024. doi:10.1177/1049732309338868.

Marovelli, B. (2019) 'Cooking and eating together in London: Food sharing initiatives as collective spaces of encounter', *Geoforum*, 99, pp. 190–201. doi:10.1016/j.geoforum.2018.09.006.

Marschner, P. (2012) *Mineral Nutrition of Higher Plants*. 3rd edn. Edited by P. Marschner. London, UK: Academic Press.

Marsden, J. *et al.* (2020) 'Mitigating and learning from the impact of COVID-19 infection on addictive disorders', *Addiction*, 115(6), pp. 1007–1010. doi:10.1111/add.15080.

Martin-Gorriz, B., Maestre-Valero, J.F., *et al.* (2021) 'Recycling drainage effluents using reverse osmosis powered by photovoltaic solar energy in hydroponic tomato production: Environmental footprint analysis. Supplementary Information, Appendix A', *Journal of Environmental Management*, 297(113326). doi:10.1016/j.jenvman.2021.113326.

Martin-Gorriz, B., Maestre-Valero, J. F., *et al.* (2021) 'Recycling drainage effluents using reverse osmosis powered by photovoltaic solar energy in hydroponic tomato production: Environmental footprint analysis', *Journal of Environmental Management*, 297(113326). doi:10.1016/j.jenvman.2021.113326.

Martin, K.S. *et al.* (2004) 'Social capital is associated with decreased risk of hunger', *Social Science & Medicine*, 58(12), pp. 2645–2654. doi:10.1016/j.socscimed.2003.09.026.

Martin, M.S. *et al.* (2016) 'Food insecurity and mental illness: Disproportionate impacts in the context of perceived stress and social isolation', *Public Health*, 132, pp. 86–91. doi:10.1016/j.puhe.2015.11.014.

Martínez-Blanco, J. *et al.* (2009) 'Life cycle assessment of the use of compost from municipal organic waste for fertilization of tomato crops', *Resources, Conservation and Recycling*, 53(6), pp. 340–351. doi:10.1016/j.resconrec.2009.02.003.

Martínez-Blanco, J. *et al.* (2010) 'The use of life cycle assessment for the comparison of biowaste composting at home and full scale', *Waste Management*, 30(6), pp. 983–994. doi:10.1016/j.wasman.2010.02.023.

Martinez, S. and Morard, P. (2000) *Recyclage des solutions nutritives en culture hors sol*. Toulouse, France: École Nationale Supérieure Agronomique de Toulouse.

Marttila, M.P. *et al.* (2021) 'Agro-industrial symbiosis and alternative heating systems for decreasing the global warming potential of greenhouse production', *Sustainability (Switzerland)*, 13(16). doi:10.3390/su13169040.

Maskell, L.C. *et al.* (2013) 'Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity', *Journal of Applied Ecology*, 50(3), pp. 561–571. doi:10.1111/1365-2664.12085.

Massa, D. *et al.* (2010) 'Strategies to decrease water drainage and nitrate emission from soilless cultures of greenhouse tomato', *Agricultural Water Management*, 97(7), pp. 971–980. doi:10.1016/j.agwat.2010.01.029.

Mathie, A. and Cunningham, G. (2003) 'From clients to citizens: Asset-based Community Development as a strategy for community-driven development', *Development in Practice*, 13(5), pp. 474–486. doi:10.1080/0961452032000125857.

Mattheys, K., Warren, J. and Bambra, C. (2018) '"Treading in sand": A qualitative study of the impact of austerity on inequalities in mental health', *Social Policy & Administration*, 52(7), pp. 1275–1289. doi:10.1111/spol.12348.

Mbah, R.E. and Wasum, D. (2022) 'Russian-Ukraine 2022 War: A Review of the Economic Impact of Russian-Ukraine Crisis on the USA, UK, Canada, and Europe', *Advances in Social Sciences Research Journal*, 9(3), pp. 144–153. doi:10.14738/assrj.93.12005.

Mbow, C. *et al.* (2019) 'Food security', in P.R. Shukla, J. Skea, E. Calvo Buendia, V. Masson-Delmotte, H.-O. Pörtner, D.C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. Portugal Pereira, P. Vyas, E. Huntley, K. Kissick, M., J.M. (ed.) *Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems*. IPCC, pp. 437–550. Available at: https://www.ipcc.ch/site/assets/uploads/sites/4/2020/02/SRCCL-Chapter-5.pdf.

McClintock, N. (2018) 'Cultivating (a) Sustainability Capital: Urban Agriculture, Ecogentrification, and the Uneven Valorization of Social Reproduction', *Annals of the American Association of Geographers*, 108(2), pp. 579–590. doi:10.1080/24694452.2017.1365582.

McClintock, N., Cooper, J. and Khandeshi, S. (2013) 'Assessing the potential contribution of vacant land to urban vegetable production and consumption in Oakland, California', *Landscape and Urban Planning*, 111, pp. 46–58. doi:10.1016/j.landurbplan.2012.12.009.

McDougall, R., Kristiansen, P. and Rader, R. (2018) 'Small-scale urban agriculture results in high yields but requires judicious management of inputs to achieve sustainability', *Proceedings of the National Academy of Sciences*, 116(1), pp. 129–134. doi:10.1073/pnas.1809707115.

McMichael, P. (2005) 'Global Development and The Corporate Food Regime', in Buttel, F.H. and McMichael, P. (eds) *New directions in the sociology of global development*. Oxford, UK: Elsevier Ltd., pp. 265–299. doi:10.1016/S1057-1922(05)11010-5.

McMichael, P. (2009) 'A food regime analysis of the "world food crisis"', *Agriculture and Human Values*, 26, pp. 281–295. doi:10.1007/s10460-009-9218-5.

McNeill, G., Dowler, E. and Shields, K. (2022) *Inside the box: an analysis of the UK's emergency food distribution scheme*, *Global Academy of Agriculture and Food Systems*. Available at: https://www.ed.ac.uk/global-agriculture-food-systems/gaafs-news/blogs/inside-the-box-an-analysis-of-the-uk-s-emergency-f (Accessed: 14 September 2022).

McNeill, K. (2011) *Talking With Their Mouths Half Full: Food Insecurity in the Hamilton Community*. University of Waikato. Available at: https://researchcommons.waikato.ac.nz/bitstream/handle/10289/5458/thesis.pdf?seq.

Mead, B.R. *et al.* (2021) 'Growing your own in times of crisis: the role of home food growing in perceived food insecurity and well-being during the early COVID-19 lockdown', *Emerald Open Research*, 3(7). doi:10.35241/emeraldopenres.14186.2.

Meier, M.S. *et al.* (2015) 'Environmental impacts of organic and conventional agricultural products - Are the differences captured by life cycle assessment?', *Journal of Environmental Management*, 149, pp. 193–208. doi:10.1016/j.jenvman.2014.10.006.

Meisterling, K., Samaras, C. and Schweizer, V. (2009) 'Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat', *Journal of Cleaner Production*, 17(2), pp. 222–230. doi:10.1016/j.jclepro.2008.04.009.

Mekonnen, M.M., Lutter, S. and Martinez, A. (2016) 'Anthropogenic nitrogen and phosphorus emissions and related grey water footprints caused by EU-27's crop production and consumption', *Water (Switzerland)*, 8(1), pp. 1–14. doi:10.3390/w8010030.

Mels, A., Bisschops, I. and Swart, B. (2008) *Zware metalen in meststoffen – vergelijking van urine en zwart water met in Nederland toegepaste meststoffen*. Netherlands: LeAf.

Merino, J. *et al.* (2021) 'Diet quality and risk and severity of COVID-19: a prospective cohort study', *Gut*, 70(11), pp. 2096–2104. doi:10.1136/gutjnl-2021-325353.

Mesnage, R. and Antoniou, M.N. (2018) ‘Ignoring Adjuvant Toxicity Falsifies the Safety Profile of Commercial Pesticides’, *Frontiers in Public Health*, 5. doi:10.3389/fpubh.2017.00361.

Mesnage, R., Benbrook, C. and Antoniou, M.N. (2019) ‘Insight into the confusion over surfactant co-formulants in glyphosate-based herbicides’, *Food and Chemical Toxicology*, 128, pp. 137–145. doi:10.1016/j.fct.2019.03.053.

Middleton, G. *et al.* (2018) ‘The experiences and perceptions of food banks amongst users in high-income countries: An international scoping review’, *Appetite*, 120, pp. 698–708. doi:10.1016/j.appet.2017.10.029.

Migliorini, P. and Wezel, A. (2017) ‘Converging and diverging principles and practices of organic agriculture regulations and agroecology. A review’, *Agronomy for Sustainable Development*, 37(6). doi:10.1007/s13593-017-0472-4.

Milbourne, P. and Coulson, H. (2021) ‘Migrant labour in the UK’s post-Brexit agri-food system: Ambiguities, contradictions and precarities’, *Journal of Rural Studies*, 86, pp. 430–439. doi:10.1016/j.jrurstud.2021.07.009.

Millennium Ecosystem Assessment (2003) *Ecosystems and Human Well-being*. Washington DC: Island Press. doi:10.1196/annals.1439.003.

Mitchell, R.G. *et al.* (2014) ‘Lead (Pb) and other metals in New York City community garden soils: Factors influencing contaminant distributions’, *Environmental Pollution*, 187, pp. 162–169. doi:10.1016/j.envpol.2014.01.007.

Moghaddam, A. (2006) ‘Coding issues in grounded theory’, *Issues in educational research*, 16(1), pp. 52–66.

Mohammed Abdul, K.S. *et al.* (2015) ‘Arsenic and human health effects: A review’, *Environmental Toxicology and Pharmacology*, 40(3), pp. 828–846. doi:10.1016/j.etap.2015.09.016.

Mok, H.F. *et al.* (2014) ‘Strawberry fields forever? Urban agriculture in developed countries: A review’, *Agronomy for Sustainable Development*, 34(1), pp. 21–43. doi:10.1007/s13593-013-0156-7.

Molden, D. (2007) *Water for Food, Water for Life: A Comprehensive Assessment of Water Management in Agriculture*. London, UK: International Water Management Institute.

Molodynski, A. *et al.* (2021) ‘What does COVID mean for UK mental health care?’, *International Journal of Social Psychiatry*, 67(7), pp. 823–825. doi:10.1177/0020764020932592.

Montero, J.I. *et al.* (2011) *Environmental and economic profile of present greenhouse production systems in Europe*. Wageningen. Available at: https://library.wur.nl/WebQuery/wurpubs/fulltext/222832.

Monto, M., Ganesh, L.S. and Varghese, K. (2005) *Sustainability and Human Settlements: Fundamental Issues, Modeling and Simulations*. Thousand Oaks, CA, USA: Sage Publications.

Morton, L.W. *et al.* (2005) ‘Solving the Problems of Iowa Food Deserts: Food Insecurity and Civic Structure’, *Rural Sociology*, 70(1), pp. 94–112. doi:10.1526/0036011053294628.

Mosier, A. *et al.* (1998) ‘Closing the global N2O budget: nitrous oxide emissions through the agricultural nitrogen cycle’, *Nutrient Cycling in Agroecosystems*, 52, pp. 225–248. doi:10.1023/A:1009740530221.

Mougeot, L.J.A. (2005) *Agropolis: The Social, Political, and Environmental Dimensions of Urban Agriculture*. London, UK: Earthscan, International Development Research Centre.

Mould, O. *et al.* (2022) ‘Solidarity, not charity: Learning the lessons of the COVID-9 pandemic to reconceptualise the radicality of mutual aid', *Transactions of the Institute of British Geographers*, 47(4), pp. 866–879. doi:10.1111/tran.12553.

Mouron, P. *et al.* (2017) *World Food LCA Database modelling documentation for crop production*. Lausanne and Zurich, Switzerland: Quantis and Agroscope.

Mukherjee, A. and Lal, R. (2014) 'Comparison of soil quality index using three methods', *PLoS ONE*, 9(8). doi:10.1371/journal.pone.0105981.

Muller, A., Ferré, M., *et al.* (2017) 'Can soil-less crop production be a sustainable option for soil conservation and future agriculture?', *Land Use Policy*, 69, pp. 102–105. doi:10.1016/j.landusepol.2017.09.014.

Muller, A., Schader, C., *et al.* (2017) 'Strategies for feeding the world more sustainably with organic agriculture', *Nature Communications*, 8(1), pp. 1–13. doi:10.1038/s41467-017-01410-w.

Mullins, L. *et al.* (2021) 'Home Food Gardening in Canada in Response to the COVID-19 Pandemic', *Sustainability*, 13(6), p. 3056. doi:10.3390/su13063056.

Muluneh, M.G. (2021) 'Impact of climate change on biodiversity and food security: a global perspective—a review article', *Agriculture and Food Security*, 10(1), pp. 1–25. doi:10.1186/s40066-021-00318-5.

Muñoz, I., Milà I Canals, L. and Fernández-Alba, A.R. (2010) 'Life cycle assessment of the average Spanish diet including human excretion', *International Journal of Life Cycle Assessment*, 15(8), pp. 794–805. doi:10.1007/s11367-010-0188-z.

Muralikrishna, I. V. and Manickam, V. (2017) 'Life Cycle Assessment', in Muralikrishna, I. V. and Manickam, V. (eds) *Environmental Management: Science and Engineering for Industry*. Elsevier, pp. 57–75. doi:10.1016/B978-0-12-811989-1.00005-1.

Myers, C.A. (2020) 'Food Insecurity and Psychological Distress: a Review of the Recent Literature', *Current Nutrition Reports*, 9(2), pp. 107–118. doi:10.1007/s13668-020-00309-1.

Naik, A. (2017) 'In search of farmer wellbeing', *International Journal of Agricultural Management*, 6(1), pp. 1–3. doi:10.5836/ijam/2017-06-01.

Nakagawa, Y. and Shaw, R. (2004) 'Social capital: A missing link to disaster recovery', *International Journal of Mass Emergencies and Disasters*, 22(1), pp. 5–34. doi:10.1177/028072700402200101.

Nasiadek, M. *et al.* (2020) 'The Role of Zinc in Selected Female Reproductive System Disorders', *Nutrients*, 12(8), p. 2464. doi:10.3390/nu12082464.

National Audit Office (2020) *Universal Credit: getting to first payment*. Available at: https://www.nao.org.uk/wp-content/uploads/2020/07/Universal-Credit-getting-to-first-payment.pdf.

Natural Capital Committee (2019) *Advice on soil management*. London, UK. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/909069/ncc-advice-soil-management.pdf.

Natural Environment Research Council (2017) *Land Cover Map 2015 Dataset Documentation*. Wallingford, UK. Available at: https://www.ceh.ac.uk/services/land-cover-map-2015.

Navarro, A. and López-Bao, J.V. (2018) 'Towards a greener Common Agricultural Policy', *Nature Ecology and Evolution*, 2(12), pp. 1830–1833. doi:10.1038/s41559-018-0724-y.

Neff, R.A., Chan, I.L. and Smith, K.C. (2009) 'Yesterday's dinner, tomorrow's weather, today's news? US newspaper coverage of food system contributions to climate change', *Public Health Nutrition*, 12(7), pp. 1006–1014. doi:10.1017/S1368980008003480.

Nelson, C. and Stroink, M. (2014) 'Accessibility and Viability: A Complex Adaptive Systems

Approach to a Wicked Problem for the Local Food Movement', *Journal of Agriculture, Food Systems, and Community Development*, 4(4), pp. 1–16. doi:10.5304/jafscd.2014.044.016.

Nemecek, T. *et al.* (2011) 'Life cycle assessment of Swiss farming systems: I. Integrated and organic farming', *Agricultural Systems*, 104(3), pp. 217–232. doi:10.1016/j.agsy.2010.10.002.

Nemecek, T. *et al.* (2015) *World Food LCA Database: Methodological Guidelines for the Life Cycle Inventory of Agricultural Products, Version 3.0*. Lausanne and Zurich, Switzerland: Quantis and Agroscope.

Nemecek, T. *et al.* (2016) 'Environmental impacts of food consumption and nutrition: where are we and what is next?', *International Journal of Life Cycle Assessment*, 21(5), pp. 607–620. doi:10.1007/s11367-016-1071-3.

Nemecek, T. *et al.* (2019) *World Food LCA Database: Methodological Guidelines for the Life Cycle Inventory of Agricultural Products, Version 3.5*. Lausanne and Zurich, Switzerland: Quantis and Agroscope.

Nemecek, T. and Kagi, T. (2007) *Life cycle inventories of Agricultural Production Systems. Data v2.0*. Zürich and Dübendorf, Switzerland: ecoinvent Centre. Available at: http://www.upe.poli.br/~cardim/PEC/Ecoinvent LCA/ecoinventReports/15_Agriculture.pdf.

Nemecek, T. and Schnetzer, J. (2011) *Methods of assessment of direct field emissions for LCIs of agricultural production systems. Data v3.0, Agroscope Reckenholz-Tänikon Research Station ART*. Zurich, Switzerland: Agroscope.

Nemecek, T., Schnetzer, J. and Reinhard, J. (2016) 'Updated and harmonised greenhouse gas emissions for crop inventories', *The International Journal of Life Cycle Assessment*, 21(9), pp. 1361–1378. doi:10.1007/s11367-014-0712-7.

Newbold, T. *et al.* (2015) 'Global effects of land use on local terrestrial biodiversity', *Nature*, 520(7545), pp. 45–50. doi:10.1038/nature14324.

NFS (2021a) *National Food Service: The NFS Guide*. Available at: https://nationalfoodservice.uk/guides.

NFS (2021b) *National Food Service support COVID-19*. Available at: https://nationalfoodservice.uk/nfs-covid-19-edm (Accessed: 31 January 2023).

NFU (2019) *Achieving Net Zero - Farming's 2040 Goal*. Stoneleigh, UK: National Farmers' Union. Available at: https://www.nfuonline.com/nfu-online/business/regulation/achieving-net-zero-farmings-2040-goal/.

NFU (2021a) *British food: leading the way*. Stoneleigh Park, UK: National Farmers' Union. Available at: https://www.nfuonline.com/media/s4xluxgg/british-food-leading-the-way.pdf.

NFU (2021b) *Our environment forum chair's view on SFI and ELMs, National Farmers' Union*. Available at: https://www.nfuonline.com/updates-and-information/nfu-environment-forum-chair-s-view-on-sfi-and-elms/ (Accessed: 7 January 2023).

NFU (2021c) *Sustainable Farming Incentive brings significant costs, National Farmers' Union*. Available at: https://www.nfuonline.com/updates-and-information/sustainable-farming-incentive-brings-significant-costs/ (Accessed: 7 January 2023).

Nicholls, R.J. and Lowe, J.A. (2004) 'Benefits of mitigation of climate change for coastal areas', *Global Environmental Change*, 14(3), pp. 229–244. doi:10.1016/j.gloenvcha.2004.04.005.

Nicholson, F.A. *et al.* (2003) 'An inventory of heavy metals inputs to agricultural soils in England and Wales', *Science of the Total Environment*, 311(1–3), pp. 205–219. doi:10.1016/S0048-9697(03)00139-6.

NOAA (2022) *Climate at a Glance, National Oceanic and Atmospheric Administration: Climate Monitoring*. Available at: https://www.ncei.noaa.gov/access/monitoring/climate-at-a-glance/county/mapping/9/pcp/201912/12/value (Accessed: 18 August 2022).

Noël, J.F. and O'Connor, M. (1998) 'Strong sustainability: towards indicators for sustainability of critical natural capital', in Faucheux, S. and O'Connor, M. (eds) *Valuation for Sustainable Development: Methods and Policy Indicators*. Cheltenham, UK: Edward Elgar, pp. 75–97.

Nogawa, K. *et al.* (2004) 'Environmental cadmium exposure, adverse effects and preventive measures in Japan', *BioMetals*, 17(5), pp. 581–587. doi:10.1023/B:BIOM.0000045742.81440.9c.

Nolan, B.T., Hitt, K.J. and Ruddy, B.C. (2002) 'Probability of nitrate contamination of recently recharged groundwaters in the conterminous United States', *Environmental Science and Technology*, 36(10), pp. 2138–2145. doi:10.1021/es0113854.

Noonan, K., Corman, H. and Reichman, N.E. (2016) 'Effects of maternal depression on family food insecurity', *Economics & Human Biology*, 22, pp. 201–215. doi:10.1016/j.ehb.2016.04.004.

Norton, G.J. *et al.* (2015) 'Cadmium and lead in vegetable and fruit produce selected from specific regional areas of the UK', *Science of The Total Environment*, 533, pp. 520–527. doi:10.1016/j.scitotenv.2015.06.130.

Norton, L. *et al.* (2016) 'The importance of scale in the development of ecosystem service indicators?', *Ecological Indicators*, 61, pp. 130–140. doi:10.1016/j.ecolind.2015.08.051.

Notarnicola, B. *et al.* (2015) 'Life Cycle Assessment in the agri-food sector: an overview of its key aspects, international initiatives, certification, labelling schemesand methodological issues', in Notarnicola, B. et al. (eds) *Life Cycle Assessment in the Agri-food Sector*. 1st edn. Cham: Springer, pp. 1–56. doi:10.1007/978-3-319-11940-3_1.

Notarnicola, B. *et al.* (2017) 'Environmental impacts of food consumption in Europe', *Journal of Cleaner Production*, 140, pp. 753–765. doi:10.1016/j.jclepro.2016.06.080.

Nowack, B. *et al.* (2001) 'Elevated Lead and Zinc Contents in Remote Alpine Soils of the Swiss National Park', *Journal of Environmental Quality*, 30(3), pp. 919–926. doi:10.2134/jeq2001.303919x.

NRCS (2021) *Web Soil Survey, United States Department of Agriculture, Natural Resources Conservation Service*. Available at: https://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx (Accessed: 16 August 2022).

Ntontis, E. *et al.* (2022) 'Tracking the nature and trajectory of social support in Facebook mutual aid groups during the COVID-19 pandemic', *International Journal of Disaster Risk Reduction*, 76(February), p. 103043. doi:10.1016/j.ijdrr.2022.103043.

O'Carroll, L. (2020) 'British workers reject fruit-picking jobs as Romanians flown in', *The Guardian*, 17 April. Available at: https://www.theguardian.com/environment/2020/apr/17/british-workers-reject-fruit-picking-jobs-as-romanians-flown-in-coronavirus.

O'Connor, R.C. *et al.* (2021) 'Mental health and well-being during the COVID-19 pandemic: Longitudinal analyses of adults in the UK COVID-19 Mental Health & Wellbeing study', *British Journal of Psychiatry*, 218(6), pp. 326–333. doi:10.1192/bjp.2020.212.

O'Hara, M. (2015) *Austerity Bites: A Journey to the Sharp End of Cuts in the UK*. Bristol, UK: Policy Press.

O'Hara, S. and Ivanic, M. (2022) 'Food Security and Lifestyle Vulnerabilities as Systemic Influencers of COVID-19 Survivability', *Medical Research Archives*, 10(8), pp. 1–12. doi:10.18103/mra.v10i8.2989.

O'Sullivan, R. (2015) *American Organic: A Cultural History of Farming, Gardening, Shopping, and Eating*. Lawrence, USA: University of Kansas Press.

Oberholzer, H.-R. *et al.* (2006) *SALCA Soil Quality: Methode zur Beurteilung der Wirkungen landwirtschaftlicher Bewirtschaftung auf die Bodenqualität in Ökobilanzen SALCA-SQ*. Agroscope FAL Reckenholz. Available at: https://www.agroscope.admin.ch/agroscope/en/home/topics/environment-resources/life-cycle-assessment/life-cycle-assessment-methods/life-cycle-assessment-method-salca.html#-654693770.

Oberndorfer, E. *et al.* (2007) 'Green Roofs as Urban Ecosystems: Ecological Structures, Functions, and Services', *BioScience*, 57(10), pp. 823–833. doi:10.1641/B571005.

Oehl, F. *et al.* (2002) 'Phosphorus budget and phosphorus availability in soils under organic and conventional farming', *Nutrient Cycling in Agroecosystems*, 62(1), pp. 25–35. doi:10.3929/ethz-b-000422918.

Office for National Statistics (2018) *Labour in the agriculture industry, UK: February 2018*. Available at: https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/internationalmigration/articles/labourintheagricultureindustry/2018-02-06 (Accessed: 11 October 2022).

Office for National Statistics (2019) *UK natural capital: urban accounts*. Available at: https://www.ons.gov.uk/economy/environmentalaccounts/bulletins/uknaturalcapital/urbanaccounts (Accessed: 31 January 2023).

Office for National Statistics (2021) *Personal well-being in the UK, quarterly: April 2011 to September 2020*. London, UK. Available at: https://www.ons.gov.uk/peoplepopulationandcommunity/wellbeing/bulletins/personalwellbeingintheukquarterly/april2011toseptember2020 (Accessed: 31 January 2023).

OHCHR (2023) *Status of Ratification Interactive Dashboard: International Covenant on Economic, Social and Cultural Rights, Office of the United Nations High Commissioner for Human Rights*. Available at: https://indicators.ohchr.org/ (Accessed: 19 January 2023).

Okereafor, U. *et al.* (2020) 'Toxic metal implications on agricultural soils, plants, animals, aquatic life and human health', *International Journal of Environmental Research and Public Health*, 17(7), pp. 1–24. doi:10.3390/ijerph17072204.

Oldenburg, R. (1989) *The great good place: Cafés, coffee shops, community centers, beauty parlors, general stores, bars, hangouts, and how they get you through the day*. Paragon House Publishers.

Oliver, M.A. and Gregory, P.J. (2015) 'Soil, food security and human health: A review', *European Journal of Soil Science*, 66(2), pp. 257–276. doi:10.1111/ejss.12216.

Opitz, I. *et al.* (2016) 'Contributing to food security in urban areas: differences between urban agriculture and peri-urban agriculture in the Global North', *Agriculture and Human Values*, 33, pp. 341–358. doi:10.1007/s10460-015-9610-2.

Organic Farmers & Growers (2019) *Organic Certification*. Available at: https://ofgorganic.org/certification/organic-certification (Accessed: 20 December 2019).

OriginLab (2022) 'Origin(Pro) v.2022b'. Northampton, MA, USA: OriginLab Corporation. Available at: https://www.originlab.com/.

Ostrom, E. (1990) *Governing the commons: The evolution of institutions for collective action*. New York, NY, USA: Cambridge University Press.

Otero, N. *et al.* (2005) 'Fertiliser characterisation: Major, trace and rare earth elements', *Applied Geochemistry*, 20(8), pp. 1473–1488. doi:10.1016/j.apgeochem.2005.04.002.

Pachankis, J.E. (2007) 'The psychological implications of concealing a stigma: A cognitive-affective-behavioral model', *Psychological Bulletin*, 133(2), pp. 328–345. doi:10.1037/0033-2909.133.2.328.

Palm, C. *et al.* (2014) 'Conservation agriculture and ecosystem services: An overview', *Agriculture,* *Ecosystems & Environment*, 187, pp. 87–105. doi:10.1016/j.agee.2013.10.010.

Pardon, L. *et al.* (2016) 'Quantifying nitrogen losses in oil palm plantations: Models and challenges', *Biogeosciences*, 13(19), pp. 5433–5452. doi:10.5194/bg-13-5433-2016.

Parsons, K., Hawkes, C. and Wells, R. (2019) 'Brief 2. What is the food system? A food policy perspective.', in *Rethinking Food Policy: A Fresh Approach to Policy and Practice*. London, UK: Centre for Food Policy, pp. 1–8.

Patrick, R. and Lee, T. (2021) *Advance to debt: Paying back benefit debt - what happens when deductions are made to benefit payments?* Covid Realities. Available at: https://covidrealities.org/learnings/write-ups/debt-deductions.

Patzel, N., Sticher, H. and Karlen, D.L. (2000) 'Soil Fertility - Phenomenon and Concept', *Journal of Plant Nutrition and Soil Science*, 163(2), pp. 129–142. doi:10.1002/(SICI)1522-2624(200004)163:2<129::AID-JPLN129>3.0.CO;2-D.

Payen, S., Basset-Mens, C. and Perret, S. (2015) 'LCA of local and imported tomato: An energy and water trade-off', *Journal of Cleaner Production*, 87(1), pp. 139–148. doi:10.1016/j.jclepro.2014.10.007.

Pe'er, G. *et al.* (2014) 'EU agricultural reform fails on biodiversity', *Science*, 344(6188), pp. 1090–1092. doi:10.1126/science.1253425.

Pearson, L.J., Pearson, L. and Pearson, C.J. (2010) 'Sustainable urban agriculture: stocktake and opportunities', *International Journal of Agricultural Sustainability*, 8(1–2), pp. 7–19. doi:10.3763/ijas.2009.0468.

Pemberton, S. *et al.* (2017) 'Endless Pressure: Life on a Low Income in Austere Times', *Social Policy and Administration*, 51(7), pp. 1156–1173. doi:10.1111/spol.12233.

Peoples, C.E. (2015) *Bowling alone but eating together: Exploring the role of gathering places as community social capital in small towns*. Iowa State University.

Perrin, A., Basset-Mens, C. and Gabrielle, B. (2014) 'Life cycle assessment of vegetable products: A review focusing on cropping systems diversity and the estimation of field emissions', *International Journal of Life Cycle Assessment*, 19(6), pp. 1247–1263. doi:10.1007/s11367-014-0724-3.

Perry, J. *et al.* (2014) *Emergency Use Only: Understanding and reducing the use of food banks in the UK*. London, UK: Child Poverty Action Group, The Church of England, Oxfam and The Trussell Trust. Available at: http://www.cpag.org.uk/sites/default/files/Foodbank Report_web.pdf.

Peters, B.G. and Pierre, J. (2014) 'Food Policy as a Wicked Problem: Contending with Multiple Demands and Actors', *World Food Policy*, 1(1), pp. 4–11. doi:10.18278/wfp.1.1.1.

Petetin, L. (2020) 'The COVID-19 crisis: An opportunity to integrate food democracy into post-pandemic food systems', *European Journal of Risk Regulation*, 11(2), pp. 326–336. doi:10.1017/err.2020.40.

Pew Research Center (2016) *The New Food Fights: U.S. Public Divides Over Food Science*. Available at: https://www.pewresearch.org/science/2016/12/01/the-new-food-fights/.

Pfister, S. *et al.* (2011) 'Environmental Impacts of Water Use in Global Crop Production: Hotspots and Trade-Offs with Land Use', *Environmental Science & Technology*, 45(13), pp. 5761–5768. doi:10.1021/es1041755.

Phalan, B. *et al.* (2011) 'Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared', *Science*, 333(6047), pp. 1289–1291. doi:10.1126/science.1208742.

Pierce, M. *et al.* (2020) 'Mental health before and during the COVID-19 pandemic: a longitudinal probability sample survey of the UK population', *The Lancet Psychiatry*, 7(10), pp. 883–892.

doi:10.1016/S2215-0366(20)30308-4.

Pinamonti, F. *et al.* (1997) ‘The use of compost: its effects on heavy metal levels in soil and plants’, *Resources, Conservation and Recycling*, 21(2), pp. 129–143. doi:10.1016/S0921-3449(97)00032-3.

Pingali, P.L. (2012) ‘Green Revolution: Impacts, limits, and the path ahead’, *Proceedings of the National Academy of Sciences*, 109(31), pp. 12302–12308. doi:10.1073/pnas.0912953109.

van der Ploeg, J.D. (2010) ‘The Food Crisis, Industrialized Farming and the Imperial Regime’, *Journal of Agrarian Change*, 10(1), pp. 98–106. doi:10.1111/j.1471-0366.2009.00251.x.

Pluimers, J.C. *et al.* (2000) ‘Quantifying the environmental impact of production in agriculture and horticulture in The Netherlands: Which emissions do we need to consider?’, *Agricultural Systems*, 66(3), pp. 167–189. doi:10.1016/S0308-521X(00)00046-9.

Pölling, B. *et al.* (2017) ‘Business models in urban farming: A comparative analysis of case studies from Spain, Italy and Germany’, *Moravian Geographical Reports*, 25(3), pp. 166–180. doi:10.1515/mgr-2017-0015.

De Ponti, T., Rijk, B. and Van Ittersum, M.K. (2012) ‘The crop yield gap between organic and conventional agriculture’, *Agricultural Systems*, 108, pp. 1–9. doi:10.1016/j.agsy.2011.12.004.

Pool, U. and Dooris, M. (2022) ‘Prevalence of food security in the UK measured by the Food Insecurity Experience Scale’, *Journal of Public Health*, 44(3), pp. 634–641. doi:10.1093/pubmed/fdab120.

Poore, J. and Nemecek, T. (2018a) ‘Reducing food’s environmental impacts through producers and consumers’, *Science*, 360(6392), pp. 987–992. doi:10.1126/science.aaq0216.

Poore, J. and Nemecek, T. (2018b) ‘Supplementary materials. Reducing food’s environmental impacts through producers and consumers’, *Science*, 360(6392), pp. 987–992. doi:10.1126/science.aaq0216.

Popkin, B.M. *et al.* (2020) ‘Individuals with obesity and COVID-19: A global perspective on the epidemiology and biological relationships’, *Obesity Reviews*, 21(11). doi:10.1111/obr.13128.

Pourmotabbed, A. *et al.* (2020) ‘Food insecurity and mental health: A systematic review and meta-analysis’, *Public Health Nutrition*, 23(10), pp. 1778–1790. doi:10.1017/S136898001900435X.

Pouyat, R.V. *et al.* (2010) ‘Chemical, physical and biological characteristics of urban soils’, in Aitkenhead, J. and Volder, A. (eds) *Urban Ecosystem Ecology*. Madison, WI, USA: Soil Science Society of America, pp. 119–152.

Power, A.G. (2010) ‘Ecosystem services and agriculture: Tradeoffs and synergies’, *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1554), pp. 2959–2971. doi:10.1098/rstb.2010.0143.

Power, M. *et al.* (2018) ‘Hidden hunger? Experiences of food insecurity amongst Pakistani and white British women’, *British Food Journal*, 120(11), pp. 2716–2732. doi:10.1108/BFJ-06-2018-0342.

Power, M. *et al.* (2020) ‘How COVID-19 has exposed inequalities in the UK food system: The case of UK food and poverty’, *Emerald Open Research*, 2, p. 11. doi:10.35241/emeraldopenres.13539.2.

Powlson, D.S. *et al.* (2011) ‘Soil management in relation to sustainable agriculture and ecosystem services’, *Food Policy*, 36(SUPPL. 1), pp. S72–S87. doi:10.1016/j.foodpol.2010.11.025.

Pozzer, A. *et al.* (2017) ‘Impact of agricultural emission reductions on fine-particulate matter and public health’, *Atmospheric Chemistry and Physics*, 17(20), pp. 12813–12826. doi:10.5194/acp-17-12813-2017.

PRé Sustainability (2022) ‘SimaPro software version 9.4.0.2’. Amersfoot, Netherlands: PRé

Sustainability. Available at: https://simapro.com/.

Pretty, J., Morrison, J.I.L. and Hine, R.E. (2003) ‘Reducing Food Poverty by Increasing Agricultural Sustainability in Developing Countries’, *Agriculture, Ecosystems and Environment*, 95(1), pp. 217–234. doi:10.1016/S0167-8809(02)00087-7.

Primas, A. (2007) *Life Cycle Inventories of new CHP systems. ecoinvent report No. 20*. Dübendorf and Zurich, Switzerland: Swiss Centre for Life Cycle Inventories. Available at: https://db.ecoinvent.org/reports/20_SmallCombinedHeatPower.pdf?area=463ee7e58cbf8.

Proksch, G. (2017) *Creating Urban Agricultural Systems – An Integrated Approach to Design*. New York, NY, USA: Taylor & Francis.

Public Health England (2021) *McCance and Widdowson’s composition of foods integrated dataset*. Available at: https://www.gov.uk/government/publications/composition-of-foods-integrated-dataset-cofid.

Purdam, K., Garratt, E.A. and Esmail, A. (2016) ‘Hungry? Food Insecurity, Social Stigma and Embarrassment in the UK’, *Sociology*, 50(6), pp. 1072–1088. doi:10.1177/0038038515594092.

Putnam, R. (2000) *Bowling Alone: The Collapse and Revival of American Community*. New York, NY, USA: Simon & Schuster.

Putnam, R.D. (1995) ‘Bowling Alone: America’s Declining Social Capital’, *Journal of Democracy*, 6(1), pp. 65–78. doi:10.1353/jod.1995.0002.

QSR International Pty Ltd. (2018) *NVivo (Version 12)*. Available at: https://www.qsrinternational.com/nvivo-qualitative-data-analysis-software/home.

Quantis and Agroscope (2019) *World Food LCA Database (WFLDB) v.3.5*. Lausanne and Zurich, Switzerland. Available at: https://quantis.com/who-we-guide/our-impact/sustainability-initiatives/wfldb-food/.

Rajan, K. *et al.* (2010) ‘Soil organic carbon—the most reliable indicator for monitoring land degradation by soil erosion’, *Current Science*, 99(6), pp. 823–827.

Reap, J. *et al.* (2008) ‘A survey of unresolved problems in life cycle assessment’, *The International Journal of Life Cycle Assessment*, 13(5), pp. 374–388. doi:10.1007/s11367-008-0009-9.

Recycled Organics Unit (2007) *Life cycle inventory and life cycle assessment for windrow composting systems*. 2nd edn. Sydney, Australia: Recycled Organics Unit and New South Wales Department of Environment and Conservation. Available at: http://www.mncompostingcouncil.org/uploads/1/5/6/0/15602762/__composting_life_cycle.pdf.

Redman, G. (2019) *The John Nix Pocketbook for Farm Management: 49th Edition for 2019*. 49th edn. Melton Mowbray, UK: Agro Business Consultants Ltd.

Reece, J. (2018) ‘Seeking Food Justice and a Just City through Local Action in Food Systems: Opportunities, Challenges, and Transformation’, *Journal of Agriculture, Food Systems, and Community Development*, 8(Suppl 2), pp. 211–215. doi:10.5304/jafscd.2018.08b.012.

Reeves, A. *et al.* (2013) ‘Austere or not? UK coalition government budgets and health inequalities’, *Journal of the Royal Society of Medicine*, 106(11), pp. 432–436. doi:10.1177/0141076813501101.

Reeves, D.W. (1994) ‘Cover Crops and Rotations’, in Hatfield, J.L. and Stewart, B.A. (eds) *Crops Residue Management*. 1st edn. Boca Raton, FL, USA: CRC Press, pp. 125–172. doi:10.1201/9781351071246.

Reganold, J.P. and Wachter, J.M. (2016) ‘Organic agriculture in the twenty-first century’, *Nature Plants*, 2(2), pp. 1–8. doi:10.1038/NPLANTS.2015.221.

Rehberger, M. and Hiete, M. (2020) 'Allocation of environmental impacts in circular and cascade use of resources-Incentive-driven allocation as a prerequisite for cascade persistence', *Sustainability (Switzerland)*, 12(11). doi:10.3390/su12114366.

Reisman, D. (2004) 'Richard Titmuss: welfare as good conduct', *European Journal of Political Economy*, 20(3), pp. 771–794. doi:10.1016/j.ejpoleco.2003.04.002.

Riches, G. and Silvasti, T. (2014) *First World Hunger Revisited*. London: Palgrave Macmillan UK. doi:10.1057/9781137298737.

Richner, W. *et al.* (2014) *Modell zur Beurteilung des Nitratauswaschungspotenzials in Ökobilanzen - SALCA- NO3. Unter Berücksichtigung der Bewirtschaftung (Fruchtfolge, Bodenbearbeitung, N-Düngung), der mikrobiellen Nitratbildung im Boden, der Stickstoffaufnahme durch die Pflanzen*. Zurich: Agroscope FAL Reckenholz. Available at: https://ira.agroscope.ch/de-CH/publication/33731.

Ridley, M. *et al.* (2020) 'Poverty, depression, and anxiety: Causal evidence and mechanisms', *Science*, 370(6522). doi:10.1126/science.aay0214.

Rinot, O. *et al.* (2019) 'Soil health assessment: A critical review of current methodologies and a proposed new approach', *Science of the Total Environment*, 648, pp. 1484–1491. doi:10.1016/j.scitotenv.2018.08.259.

Rivington, M. *et al.* (2021) 'UK food and nutrition security during and after the COVID-19 pandemic', *Nutrition Bulletin*, 46(1), pp. 88–97. doi:10.1111/nbu.12485.

Robbins, M.J. (2015) 'Exploring the "localisation" dimension of food sovereignty', *Third World Quarterly*, 36(3), pp. 449–468. doi:10.1080/01436597.2015.1024966.

Robertshaw, D. *et al.* (2022) 'Welfare at a (Social) Distance: accessing social security and employment support during COVID-19 and its aftermath', in Garthwaite, K. et al. (eds) *COVID-19 Collaborations: Researching Poverty and Low-Income Family Life during the Pandemic*. Bristol, UK: Policy Press, pp. 30–43.

Robertson, P.G. *et al.* (2014) 'Farming for Ecosystem Services: An Ecological Approach to Production Agriculture', *BioScience*, 64(5), pp. 404–415. doi:10.1093/biosci/biu037.

Robinson, D.A. *et al.* (2012) 'Natural Capital, Ecosystem Services, and Soil Change: Why Soil Science Must Embrace an Ecosystems Approach', *Vadose Zone Journal*, 11(1). doi:10.2136/vzj2011.0051.

Robinson, D.A. *et al.* (2013) 'Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation', *Soil Biology and Biochemistry*, 57, pp. 1023–1033. doi:10.1016/j.soilbio.2012.09.008.

Rochette, P. and Janzen, H.H. (2005) 'Towards a Revised Coefficient for Estimating N2O Emissions from Legumes', *Nutrient Cycling in Agroecosystems*, 73(2–3), pp. 171–179. doi:10.1007/s10705-005-0357-9.

Rodman, S.O. *et al.* (2014) '"They Just Say Organic Food Is Healthier": Perceptions of Healthy Food among Supermarket Shoppers in Southwest Baltimore', *Culture, Agriculture, Food and Environment*, 36(2), pp. 83–92. doi:10.1111/cuag.12036.

Rodriguez-Llanes, J.M., Vos, F. and Guha-Sapir, D. (2013) 'Measuring psychological resilience to disasters: Are evidence-based indicators an achievable goal?', *Environmental Health: A Global Access Science Source*, 12(1), pp. 1–10. doi:10.1186/1476-069X-12-115.

Roe, B.E., Bender, K. and Qi, D. (2021) 'The Impact of COVID-19 on Consumer Food Waste', *Applied Economic Perspectives and Policy*, 43(1), pp. 401–411. doi:10.1002/aepp.13079.

Ronga, D. *et al.* (2019) 'Effects of nitrogen management on biomass production and dry matter distribution of processing tomato cropped in southern Italy', *Agronomy*, 9(12), pp. 1–16.

doi:10.3390/agronomy9120855.

Röös, E. and Karlsson, H. (2013) ‘Effect of eating seasonal on the carbon footprint of Swedish vegetable consumption’, *Journal of Cleaner Production*, 59, pp. 63–72. doi:10.1016/j.jclepro.2013.06.035.

Rose, W. and McAuley, C. (2019) ‘Poverty and its impact on parenting in the UK: Re-defining the critical nature of the relationship through examining lived experiences in times of austerity’, *Children and Youth Services Review*, 97, pp. 134–141. doi:10.1016/j.childyouth.2017.10.021.

Rosen, J.D. (2010) ‘A Review of the Nutrition Claims Made by Proponents of Organic Food’, *Comprehensive Reviews in Food Science and Food Safety*, 9(3), pp. 270–277. doi:10.1111/j.1541-4337.2010.00108.x.

Ross, G.J. (2014) *Food deserted: race, poverty, and food vulnerability in Atlanta, 1980-2010*. Georgia Institute of Technology. Available at: https://smartech.gatech.edu/handle/1853/53062.

Rossi, F. *et al.* (2008) ‘Health-promoting substances and heavy metal content in tomatoes grown with different farming techniques’, *European Journal of Nutrition*, 47(5), pp. 266–272. doi:10.1007/s00394-008-0721-z.

Rothwell, A. *et al.* (2016) ‘Environmental performance of local food: Trade-offs and implications for climate resilience in a developed city’, *Journal of Cleaner Production*, 114, pp. 420–430. doi:10.1016/j.jclepro.2015.04.096.

Rowland, C.S. *et al.* (2007) *Land Cover Map 2015 (25m raster, GB)*. NERC Environmental Information Data Centre (Dataset). doi:10.5285/bb15e200-9349-403c-bda9-b430093807c7.

Roy, P. *et al.* (2009) ‘A review of life cycle assessment (LCA) on some food products’, *Journal of Food Engineering*, 90(1), pp. 1–10. doi:10.1016/j.jfoodeng.2008.06.016.

Roy, R.N. *et al.* (2003) *Assessment of soil nutrient balance. Approaches and Methodologies. Fertilizer and Plant Nutrition Bulletin 14*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Rufí-Salís, M. *et al.* (2020) ‘Recirculating water and nutrients in urban agriculture: An opportunity towards environmental sustainability and water use efficiency?’, *Journal of Cleaner Production*, 261. doi:10.1016/j.jclepro.2020.121213.

Rufí-Salís, M. *et al.* (2021) ‘Combining LCA and circularity assessments in complex production systems: the case of urban agriculture’, *Resources, Conservation and Recycling*, 166(105359). doi:10.1016/j.resconrec.2020.105359.

Rumpel, C., Chabbi, A. and Marschner, B. (2012) ‘Carbon Storage and Sequestration in Subsoil Horizons: Knowledge, Gaps and Potentials’, in Lal, R. et al. (eds) *Recarbonization of the Biosphere*. Dordrecht: Springer.

Runnels, V.E., Kristjansson, E. and Calhoun, M. (2011) ‘An Investigation of Adults’ Everyday Experiences and Effects of Food Insecurity in an Urban Area in Canada’, *Canadian Journal of Community Mental Health*, 30(1), pp. 157–172. doi:10.7870/cjcmh-2011-0011.

Saer, A. *et al.* (2013) ‘Life cycle assessment of a food waste composting system: Environmental impact hotspots’, *Journal of Cleaner Production*, 52, pp. 234–244. doi:10.1016/j.jclepro.2013.03.022.

Sage, C. (2013) ‘The interconnected challenges for food security from a food regimes perspective: Energy, climate and malconsumption’, *Journal of Rural Studies*, 29(1), pp. 71–80. doi:10.1016/j.jrurstud.2012.02.005.

Sage, C. (2014) ‘The transition movement and food sovereignty: From local resilience to global engagement in food system transformation’, *Journal of Consumer Culture*, 14(2), pp. 254–275. doi:10.1177/1469540514526281.

Saha, M. and 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, pp. 130–141. doi:10.1016/j.landurbplan.2017.04.015.

Salisbury, K. *et al.* (2018) ‘Is Local Produce Really More Expensive? A Comparison of Direct Market and Conventional Grocery Produce Pricing’, *Journal of Food Distribution Research*, 49(1), pp. 13–21. doi:10.22004/ag.econ.274599.

Sánchez-Bayo, F. and Wyckhuys, K.A.G. (2019) ‘Worldwide decline of the entomofauna: A review of its drivers’, *Biological Conservation*, 232, pp. 8–27. doi:10.1016/j.biocon.2019.01.020.

Sánchez, A. *et al.* (2015) ‘Greenhouse Gas from Organic Waste Composting: Emissions and Measurement’, in Lichtfouse, E., Schwarzbauer, J., and Robert, D. (eds) *CO2 Sequestration, Biofuels and Depollution. Environmental Chemistry for a Sustainable World*. Springer, Cham. doi:10.1007/s10311-015-0507-5.

Sandelowski, M. (2000) ‘Focus on research methods: Whatever happened to qualitative description?’, *Research in Nursing and Health*, 23(4), pp. 334–340. doi:10.1002/1098-240x(200008)23:4<334::aid-nur9>3.0.co;2-g.

Sanjuan-Delmás, D. *et al.* (2020) ‘Applying nutrient dynamics to adjust the nutrient-water balance in hydroponic crops. A case study with open hydroponic tomato crops from Barcelona’, *Scientia Horticulturae*, 261(108908). doi:10.1016/j.scienta.2019.108908.

Sanyé-Mengual, E. *et al.* (2015) ‘An environmental and economic life cycle assessment of rooftop greenhouse (RTG) implementation in Barcelona, Spain. Assessing new forms of urban agriculture from the greenhouse structure to the final product level’, *The International Journal of Life Cycle Assessment*, 20(3), pp. 350–366. doi:10.1007/s11367-014-0836-9.

SARE (2007) *Managing Cover Crops Profitably*. 3rd edn. Edited by A. Clark. College Park, MD, USA: Sustainable Agriculture Research and Education (SARE).

Satterthwaite, D., McGranahan, G. and Tacoli, C. (2010) ‘Urbanization and its implications for food and farming’, *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1554), pp. 2809–2820. doi:10.1098/rstb.2010.0136.

Säumel, I. *et al.* (2012) ‘How healthy is urban horticulture in high traffic areas? Trace metal concentrations in vegetable crops from plantings within inner city neighbourhoods in Berlin, Germany’, *Environmental Pollution*, 165, pp. 124–132. doi:10.1016/j.envpol.2012.02.019.

Sausse, C. *et al.* (2015) ‘Do the effects of crops on skylark (Alauda arvensis) differ between the field and landscape scales?’, *PeerJ*, 3(e1097). doi:10.7717/peerj.1097.

Schifferstein, H.N.J. and Oude Ophuis, P.A.M. (1998) ‘Health-related determinants of organic food consumption in The Netherlands’, *Food Quality and Preference*, 9(3), pp. 119–133. doi:10.1016/S0950-3293(97)00044-X.

Schindelbeck, R.R. *et al.* (2008) ‘Comprehensive assessment of soil quality for landscape and urban management’, *Landscape and Urban Planning*, 88(2–4), pp. 73–80. doi:10.1016/j.landurbplan.2008.08.006.

Schleicher, T. *et al.* (2018) *The Blue Angel for Stationary Room Air Conditioners - market analysis, technical developments and regulatory framework for criteria development*. Dessau-Roßlau, Germany. Available at: http://www.umweltbundesamt.de/publikationen.

Schmidhuber, J. and Tubiello, F.N. (2007) ‘Global food security under climate change’, *Proceedings of the National Academy of Sciences*, 104(50), pp. 19703–19708. doi:10.1073/pnas.0701976104.

Schoen, V. *et al.* (2021) ‘“We Have Been Part of the Response”: The Effects of COVID-19 on Community and Allotment Gardens in the Global North’, *Frontiers in Sustainable Food Systems*, 5(732641). doi:10.3389/fsufs.2021.732641.

Schrijvers, D.L., Loubet, P. and Sonnemann, G. (2016) ‘Critical review of guidelines against a systematic framework with regard to consistency on allocation procedures for recycling in LCA’, *The International Journal of Life Cycle Assessment*, 21(7), pp. 994–1008. doi:10.1007/s11367-016-1069-x.

De Schutter, O. (2014) *The transformative potential of the right to food: Report of the Special Rapporteur on the right to food*. Geneva, Switzerland: United Nations General Assembly. Available at: https://digitallibrary.un.org/record/766914?ln=en.

Schwartz, N., Buliung, R. and Wilson, K. (2019) ‘Disability and food access and insecurity: A scoping review of the literature’, *Health and Place*, 57(May 2018), pp. 107–121. doi:10.1016/j.healthplace.2019.03.011.

Scialabba, N.E.-H. and Müller-Lindenlauf, M. (2010) ‘Organic agriculture and climate change’, *Renewable Agriculture and Food Systems*, 25(2), pp. 158–169. doi:10.1017/S1742170510000116.

Scown, M.W., Brady, M. V. and Nicholas, K.A. (2020) ‘Billions in Misspent EU Agricultural Subsidies Could Support the Sustainable Development Goals’, *One Earth*, 3(2), pp. 237–250. doi:10.1016/j.oneear.2020.07.011.

Sen, A.K. (1981) *Poverty and famines: An essay on entitlement and deprivation*. Oxford, UK: Clarendon Press.

Sen, A.K. (1990) ‘Development as capability expansion’, *The community development reader*, 41, p. 58.

Sen, A.K. (1999a) *Commodities and Capabilities*. Oxford, UK: Oxford University Press.

Sen, A.K. (1999b) *Devleopment as freedom*. Oxford, UK: Open University Press.

Sena, M.M. *et al.* (2002) ‘Discrimination of management effects on soil parameters by using principal component analysis: A multivariate analysis case study’, *Soil and Tillage Research*, 67(2), pp. 171–181. doi:10.1016/S0167-1987(02)00063-6.

Seufert, V., Ramankutty, N. and Foley, J.A. (2012) ‘Comparing the yields of organic and conventional agriculture’, *Nature*, 485(7397), pp. 229–232. doi:10.1038/nature11069.

Sexton, R.J. (2012) ‘Market Power, Misconceptions, and Modern Agricultural Markets’, *American Journal of Agricultural Economics*, 95(2), pp. 209–219. doi:10.1093/ajae/aas102.

Shaffer, A. and Gottlieb, R. (2002) *The Persistence of L.A.’s Grocery Gap: The Need for a New Food Policy and Approach to Market Development*. Los Angeles, CA, USA: Center for Food and Justice, Occidental College.

Shafie, F.A. and Rennie, D. (2012) ‘Consumer Perceptions Towards Organic Food’, *Procedia - Social and Behavioral Sciences*, 49, pp. 360–367. doi:10.1016/j.sbspro.2012.07.034.

Shah, A.N. *et al.* (2017) ‘Soil compaction effects on soil health and crop productivity: an overview’, *Environmental Science and Pollution Research*, 24(11), pp. 10056–10067. doi:10.1007/s11356-017-8421-y.

Shah, V.P., Debella, D.C. and Ries, R.J. (2008) ‘Life cycle assessment of residential heating and cooling systems in four regions in the United States’, *Energy and Buildings*, 40(4), pp. 503–513. doi:10.1016/j.enbuild.2007.04.004.

Sharma, P., Chambial, S. and Shukla, K.K. (2015) ‘Lead and Neurotoxicity’, *Indian Journal of Clinical Biochemistry*, 30(1), pp. 1–2. doi:10.1007/s12291-015-0480-6.

Sharpley, A. (2000) ‘Phosphorus Availability’, in Sumner, M.E. (ed.) *Handbook of Soil Science*. Boca

Raton, FL, USA: CRC Press, pp. D18–D37.

Sharpley, A. and Moyer, B. (2000) 'Phosphorus Forms in Manure and Compost and Their Release during Simulated Rainfall', *Journal of Environmental Quality*, 29(5), pp. 1462–1469. doi:10.2134/jeq2000.00472425002900050012x.

Shaw, H.J. (2006) 'Food deserts: Towards the development of a classification', *Geografiska Annaler, Series B: Human Geography*, 88(2), pp. 231–247. doi:10.1111/j.0435-3684.2006.00217.x.

Shaw, P.E. and Newby, L. (1998) 'Sustainable Wealth Creation At The Local Level In An Age Of Globalization', *Regional Studies*, 32(9), pp. 863–871. doi:10.1080/00343409850118013.

Sheffield City Council (2021) *Food Poverty Working Group Interim Report*. Sheffield, UK. Available at: https://democracy.sheffield.gov.uk/mgConvert2PDF.aspx?ID=44100.

ShefFood (2023) *ShefFood: Sheffield's Food Partnership*. Available at: https://sheffood.org.uk/ (Accessed: 27 January 2023).

Shennan, C. *et al.* (2017) 'Organic and Conventional Agriculture: A Useful Framing?', *Annual Review of Environment and Resources*, 42, pp. 317–346. doi:10.1146/annurev-environ-110615-085750.

Shepherd, M.A., Harrison, R. and Webb, J. (2002) 'Managing soil organic matter – implications for soil structure on organic farms', *Soil Use and Management*, 18(3), pp. 284–292. doi:10.1079/SUM2002134.

Shields, R. (2019) *Britian Under Pressure to Attract More Young People Into Agriculture*, *Agricultural Recruitment Specialists*. Available at: https://www.agrirs.co.uk/blog/2019/01/britain-under-pressure-to-attract-more-young-people-into-agriculture? (Accessed: 26 January 2023).

Shildrick, T., MacDonald, R. and Furlong, A. (2016) 'Not single spies but in battalions: a critical, sociological engagement with the idea of so-called "Troubled Families"', *The Sociological Review*, 64(4). doi:10.1111/1467-954X.12425.

Shrestha, P., Small, G.E. and Kay, A. (2020) 'Quantifying nutrient recovery efficiency and loss from compost-based urban agriculture', *PLOS ONE*. Edited by J. Aherne, 15(4), p. e0230996. doi:10.1371/journal.pone.0230996.

Siegner, A., Sowerwine, J. and Acey, C. (2018) 'Does urban agriculture improve food security? Examining the nexus of food access and distribution of urban produced foods in the United States: A systematic review', *Sustainability*, 10(2988). doi:10.3390/su10092988.

Siegrist, M., Visschers, V.H.M. and Hartmann, C. (2015) 'Factors influencing changes in sustainability perception of various food behaviors: Results of a longitudinal study', *Food Quality and Preference*, 46, pp. 33–39. doi:10.1016/j.foodqual.2015.07.006.

Silberston, A. (1972) 'Economies of Scale in Theory and Practice', *The Economic Journal*, 82(325), p. 369. doi:10.2307/2229943.

Sligh, M. and Cierpka, T. (2007) 'Organic Values', in Lockeretz, W. (ed.) *Organic Farming: An International History*. Cambridge, MA, USA: CAB International.

Smaling, E.M.A. *et al.* (2008) 'From forest to waste: Assessment of the Brazilian soybean chain, using nitrogen as a marker', *Agriculture, Ecosystems and Environment*, 128(3), pp. 185–197. doi:10.1016/j.agee.2008.06.005.

Smaling, E.M.A., Stoorvogel, J.J. and Windmeijer, P.N. (1993) 'Calculating soil nutrient balances in Africa at different scales - II. District scale', *Fertilizer Research*, 35(3), pp. 237–250. doi:10.1007/BF00750642.

Small, G. *et al.* (2019) 'Excess phosphorus from compost applications in urban gardens creates potential pollution hotspots', *Environmental Research Communications*, 1(9), p. 091007. doi:10.1088/2515-7620/ab3b8c.

Smith, A. *et al.* (2001) *Waste management options and climate change*. Luxembourg: European Communities.

Smith, P. *et al.* (2007) 'Agriculture', in Metz, B. et al. (eds) *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK and New York, NY, USA: Cambridge University Press.

Smith, S.R. (2009) 'A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge', *Environment International*, 35(1), pp. 142–156. doi:10.1016/j.envint.2008.06.009.

Smith, V.H. and Glauber, J.W. (2019) 'The Future of US Farm Policy', *EuroChoices*, 18(1), pp. 42–48. doi:10.1111/1746-692X.12223.

Snyder, R.G. (2010) *Greenhouse Tomato Handbook, Publication No. 1828.* Crystal Springs, MS, USA: Mississippi State University Extension. Available at: https://extension.msstate.edu/sites/default/files/publications/publications/p1828.pdf.

Soil Association (2019) *Soil Association Certification*. Available at: https://www.soilassociation.org/certification/ (Accessed: 20 December 2019).

Soil Association (2022) *Transforming the way we eat*, *Causes & Campaigns*. Available at: https://www.soilassociation.org/causes-campaigns/better-food-for-all/transforming-the-way-we-eat/ (Accessed: 21 January 2023).

Soil Health Institute (2017) *National Soil Health Measurements to Accelerate Agricultural Transformation*. Available at: https://soilhealthinstitute.org/news-events/national-soil-health-measurements-accelerate-agricultural-transformation/ (Accessed: 8 January 2023).

Sosenko, F. *et al.* (2019) *State of Hunger: A study of poverty and food insecurity in the UK.* The Trussell Trust. Available at: https://www.stateofhunger.org/wp-content/uploads/2019/10/State-of-Hunger-Executive-Summary-Digital.pdf.

Sosenko, F., Bramley, G. and Bhattacharjee, A. (2022) 'Understanding the post-2010 increase in food bank use in England: new quasi-experimental analysis of the role of welfare policy', *BMC Public Health*, 22(1), pp. 1–10. doi:10.1186/s12889-022-13738-0.

Spade, D. (2020) 'Solidarity Not Charity', *Social Text*, 38(1), pp. 131–151. doi:10.1215/01642472-7971139.

Specht, K. *et al.* (2014) 'Urban agriculture of the future: an overview of sustainability aspects of food production in and on buildings', *Agriculture and Human Values*, 31, pp. 33–51. doi:10.1007/s10460-013-9448-4.

Srivastava, K. (2020) 'Association between COVID-19 and cardiovascular disease', *IJC Heart & Vasculature*, 29, p. 100583. doi:10.1016/j.ijcha.2020.100583.

Steffen, W. *et al.* (2015) 'Planetary boundaries: Guiding human development on a changing planet', *Science*, 347(6223). doi:10.1126/science.1259855.

Stein, A.J. and Santini, F. (2022) 'The sustainability of "local" food: a review for policy-makers', *Review of Agricultural, Food and Environmental Studies*, 103(1), pp. 77–89. doi:10.1007/s41130-021-00148-w.

Steiner, G., Geissler, B. and Schernhammer, E.S. (2019) 'Hunger and obesity as symptoms of non-sustainable food systems and malnutrition', *Applied Sciences (Switzerland)*, 9(6), pp. 1–16. doi:10.3390/app9061062.

Stephens, E.C. *et al.* (2020) 'Editorial: Impacts of COVID-19 on agricultural and food systems worldwide and on progress to the sustainable development goals', *Agricultural Systems*, 183(102873). doi:10.1016/j.agsy.2020.102873.

Stephens, R. (2019) *Agritecture Exclusive: D.C. set to become 3rd major US city to hire a director of urban agriculture*, *Agritecture*. Available at: https://www.agritecture.com/blog/2019/11/19/agritecture-exclusive-dc-set-to-become-3rd-major-us-city-to-hire-a-director-of-urban-agriculture (Accessed: 18 August 2020).

Sterrett, S.B. *et al.* (1996) 'Influence of fertilizer and sewage sludge compost on yield and heavy metal accumulation by lettuce grown in urban soils', *Environmental Geochemistry and Health*, 18(4), pp. 135–142. doi:10.1007/BF01771236.

Stewart, R.D. *et al.* (2018) 'What We Talk about When We Talk about Soil Health', *Agricultural & Environmental Letters*, 3(180033). doi:10.2134/ael2018.06.0033.

Stockdale, E.A. *et al.* (2001) 'Agronomic and environmental implications of organic farming systems', *Advances in Agronomy*, 70, pp. 261–327. doi:10.1016/s0065-2113(01)70007-7.

Stockdale, E.A. *et al.* (2002) 'Soil fertility in organic farming systems – fundamentally different?', *Soil Use and Management*, 18(3), pp. 301–308. doi:10.1079/sum2002143.

Stockmann, U. *et al.* (2013) 'The knowns, known unknowns and unknowns of sequestration of soil organic carbon', *Agriculture, Ecosystems and Environment*, 164(2013), pp. 80–99. doi:10.1016/j.agee.2012.10.001.

Story, M., Hamm, M.W. and Wallinga, D. (2009) 'Food systems and public health: Linkages to achieve healthier diets and healthier communities', *Journal of Hunger and Environmental Nutrition*, 4(3–4), pp. 219–224. doi:10.1080/19320240903351463.

Strauss, A.L. and Corbin, J.M. (1998) *Basics of qualitative research: Techniques and procedures for developing grounded theory*. 2nd edn. Thousand Oaks, CA, USA: Sage Publications.

Struijs, J. *et al.* (2011) 'Characterization factors for inland water eutrophication at the damage level in life cycle impact assessment', *International Journal of Life Cycle Assessment*, 16(1), pp. 59–64. doi:10.1007/s11367-010-0232-z.

Stubbs, M. (2019) *Agricultural Conservation in the 2018 Farm Bill*. Washington DC, USA: Congressional Research Service, Library of Congress. Available at: https://nationalaglawcenter.org/wp-content/uploads/2019/05/Ag-Law-Policy-on-the-Hill-Materials.pdf.

Suleiman, A.A. and Hoogenboom, G. (2007) 'Comparison of Priestley-Taylor and FAO-56 Penman-Monteith for Daily Reference Evapotranspiration Estimation in Georgia', *Journal of Irrigation and Drainage Engineering*, 133(2), pp. 175–182. doi:10.1061/(asce)0733-9437(2007)133:2(175).

Sullivan, D.M. *et al.* (2018) *Interpreting Compost Analyses*. Corvallis, OR, USA: Oregon State University Extension Service. Available at: https://catalog.extension.oregonstate.edu/sites/catalog/files/project/pdf/em9217.pdf.

Sullivan, S. (2011) '"Ecosystem Service Commodities" - A New Imperial Ecology? Implications for Animist Immanent Ecologies, with Deleuze and Guattari', *New Formations*, 69, pp. 111–128. doi:10.3898/newf.69.06.2010.

Sumner, J. (2011) 'Serving social justice: The role of the commons in sustainable food systems', *Studies in Social Justice*, 5(1), pp. 63–75. doi:10.26522/ssj.v5i1.992.

Sustain (2017) *Michael Gove says UK is facing 'eradication of soil fertility'*, *News*. Available at: https://www.sustainweb.org/news/oct17_gove_soil_fertility/ (Accessed: 8 January 2023).

Sustain (2021) *Farming sustainably through the SFI – what do we know, what's still missing and* *what next for farmers?*, *Blogs - Sustainable Farming Campaign*. Available at: https://www.sustainweb.org/blogs/jul21-sfi-scheme-update/ (Accessed: 7 January 2023).

Sustain and RSPB (2021) *The case for local food: building better local food systems to benefit society and nature*. London, UK. Available at: https://www.sustainweb.org/publications/the-case-for-local-food/.

Svanbäck, A. *et al.* (2019) 'Reducing agricultural nutrient surpluses in a large catchment – Links to livestock density', *Science of the Total Environment*, 648, pp. 1549–1559. doi:10.1016/j.scitotenv.2018.08.194.

Swinton, S.M. *et al.* (2007) 'Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits', *Ecological Economics*, 64(2), pp. 245–252. doi:10.1016/j.ecolecon.2007.09.020.

Tarasuk, V. *et al.* (2013) 'Chronic physical and mental health conditions among adults may increase vulnerability to household food insecurity', *Journal of Nutrition*, 143(11), pp. 1785–1793. doi:10.3945/jn.113.178483.

Tarasuk, V. and Beaton, G. (1999) 'Household food insecurity and hunger among families using food banks', *Canadian Journal of Public Health*, 90(2), pp. 109–113. doi:10.1007/BF03404112.

Tarasuk, V., Fafard St-Germain, A.-A. and Loopstra, R. (2020) 'The Relationship Between Food Banks and Food Insecurity: Insights from Canada', *VOLUNTAS: International Journal of Voluntary and Nonprofit Organizations*, 31(5), pp. 841–852. doi:10.1007/s11266-019-00092-w.

Tarasuk, V., McIntyre, L. and Li, J. (2007) 'Low-Income Women's Dietary Intakes Are Sensitive to the Depletion of Household Resources in One Month', *The Journal of Nutrition*, 137(8), pp. 1980–1987. doi:10.1093/jn/137.8.1980.

Taylor-Gooby, P. and Stoker, G. (2011) 'The coalition programme: A new vision for Britain or politics as usual?', *Political Quarterly*, 82(1), pp. 4–15. doi:10.1111/j.1467-923X.2011.02169.x.

The Food Foundation (2021) *A crisis within a crisis: The impact of Covid-19 on household food security*. London, UK. Available at: https://foodfoundation.org.uk/wp-content/uploads/2021/03/FF_Impact-of-Covid_FINAL.pdf.

The Foodhall Project (2020) *About Foodhall*. Available at: https://www.foodhallproject.org/about-foodhall (Accessed: 9 December 2020).

The Royal Society (2009) *Reaping the benefits: Science and the sustainable intensification of global agriculture*. London, UK: The Royal Society.

The Trussell Trust (2020a) *Coronavirus Response Impact Report: March to September 2020*. Available at: https://www.trusselltrust.org/wp-content/uploads/sites/2/2020/11/coronavirus-impact-report-final.pdf.

The Trussell Trust (2020b) *Lockdown, lifelines and the long haul: impact of Covid-19 on food banks in the Trussell Trust network*. Available at: https://www.trusselltrust.org/wp-content/uploads/sites/2/2020/09/the-impact-of-covid-19-on-food-banks-executive-summary.pdf.

The Trussell Trust (2020c) *UK Food Banks Report Busiest Month Ever, as Coalition Urgently Calls for Funding to Get Money into People's Pockets Quickly During Pandemic*. Available at: https://www.trusselltrust.org/2020/06/03/food-banks-busiest-month/ (Accessed: 14 September 2022).

The Trussell Trust (2022a) *Five Weeks Too Long*. Available at: https://www.trusselltrust.org/five-weeks-too-long/ (Accessed: 17 October 2022).

The Trussell Trust (2022b) *Trussell Trust data briefing on end-of-year statistics relating to use of food banks: April 2021 – March 2022*. Available at: https://www.trusselltrust.org/wp-content/uploads/sites/2/2022/04/EOY-Stats-2022-Data-Briefing.pdf.

Theurl, M.C. *et al*. (2014) 'Contrasted greenhouse gas emissions from local versus long-range tomato production', *Agronomy for Sustainable Development*, 34(3), pp. 593–602. doi:10.1007/s13593-013-0171-8.

Thilmany, D. *et al.* (2021) 'Local Food Supply Chain Dynamics and Resilience during COVID-19', *Applied Economic Perspectives and Policy*, 43(1), pp. 86–104. doi:10.1002/aepp.13121.

Thomas, G.M. (2021) 'Dis-mantling stigma: Parenting disabled children in an age of "neoliberal-ableism"', *The Sociological Review*, 69(2), pp. 451–467. doi:10.1177/0038026120963481.

Thompson, R.B. *et al.* (2013) 'Effect of N uptake concentration on nitrate leaching from tomato grown in free-draining soilless culture under Mediterranean conditions', *Scientia Horticulturae*, 150, pp. 387–398. doi:10.1016/j.scienta.2012.11.018.

Thorman, R.E. *et al.* (2020) 'Towards Country-Specific Nitrous Oxide Emission Factors for Manures Applied to Arable and Grassland Soils in the UK', *Frontiers in Sustainable Food Systems*, 4(May), pp. 1–19. doi:10.3389/fsufs.2020.00062.

Thornbush, M. (2015) 'Urban agriculture in the transition to low carbon cities through urban greening', *AIMS Environmental Science*, 2(3), pp. 852–867. doi:10.3934/environsci.2015.3.852.

Thornton, I. (1994) 'Sources and pathways of arsenic in south-west England: health implications', in Chapell, W.R., Abernathy, C.O., and Cothern, C.R. (eds) *Arsenic Exposure and Health*. Northwood: Science and Technology Letters, pp. 61–70.

Thorsen, D.E. and Lie, A. (2006) 'What is Neoliberalism?', *University of Oslo, Department of Political Science* [Preprint].

Thums, C.R., Farago, M.E. and Thornton, I. (2008) 'Bioavailability of trace metals in brownfield soils in an urban area in the UK', *Environmental Geochemistry and Health*, 30(6), pp. 549–563. doi:10.1007/s10653-008-9185-6.

Tillman, A.-M. *et al.* (1994) 'Choice of system boundaries in life cycle assessment', *Journal of Cleaner Production*, 2(1), pp. 21–29. doi:10.1016/0959-6526(94)90021-3.

Tiratelli, L. and Kaye, S. (2020) *Communities vs. coronavirus: the rise of mutual aid*. London, UK: New Local Government Network. doi:10.1093/acprof:oso/9780195336450.001.0001.

Tittonell, P. *et al.* (2021) 'Emerging responses to the COVID-19 crisis from family farming and the agroecology movement in Latin America – A rediscovery of food, farmers and collective action', *Agricultural Systems*, 190(103098). doi:10.1016/j.agsy.2021.103098.

Tomlinson, I. (2013) 'Doubling food production to feed the 9 billion: A critical perspective on a key discourse of food security in the UK', *Journal of Rural Studies*, 29, pp. 81–90. doi:10.1016/j.jrurstud.2011.09.001.

Torrellas, M., Antón, A., Ruijs, M., *et al.* (2012) 'Environmental and economic assessment of protected crops in four European scenarios', *Journal of Cleaner Production*, 28, pp. 45–55. doi:10.1016/j.jclepro.2011.11.012.

Torrellas, M., Antón, A., López, J.C., *et al.* (2012) 'LCA of a tomato crop in a multi-tunnel greenhouse in Almeria', 17, pp. 863–875. doi:10.1007/s11367-012-0409-8.

Torres Pineda, I. *et al.* (2021) 'Review of inventory data in life cycle assessment applied in production of fresh tomato in greenhouse', *Journal of Cleaner Production*, 282(124395). doi:10.1016/j.jclepro.2020.124395.

Tovey, J.P. (2008) 'Whose rights and who's right? valuing ecosystem services in Victoria, Australia', *Landscape Research*, 33(2), pp. 197–209. doi:10.1080/01426390801908426.

Trades Union Congress (2020) *Universal Credit and the impact of the five week wait for payment*.

Available at: https://www.tuc.org.uk/research-analysis/reports/universal-credit-and-impact-five-week-wait-payment.

Troitino, D.R. and Chochia, A. (2013) 'The Common Agricultural Policy, Its Role in European Integration and Influence on the Enlargements of the Organization (Case Study: Georgia)', *International and Comparative Law Review*, 13(1), pp. 39–60. doi:10.1515/iclr-2016-0057.

Tscharntke, T. *et al.* (2021) 'Beyond organic farming – harnessing biodiversity-friendly landscapes', *Trends in Ecology & Evolution*, 36(10), pp. 919–930. doi:10.1016/j.tree.2021.06.010.

Tully, K.L. and McAskill, C. (2020) 'Promoting soil health in organically managed systems: a review', *Organic Agriculture*, 10(3), pp. 339–358. doi:10.1007/s13165-019-00275-1.

Tuomisto, H.L. *et al.* (2012) 'Does organic farming reduce environmental impacts? - A meta-analysis of European research', *Journal of Environmental Management*, 112(834), pp. 309–320. doi:10.1016/j.jenvman.2012.08.018.

Turcu, C. and Rotolo, M. (2022) 'Disrupting from the ground up: community-led and place-based food governance in London during COVID-19', *Urban Governance*, 2(1), pp. 178–187. doi:10.1016/j.ugj.2022.04.006.

Turner, B. (2011) 'Embodied connections: Sustainability, food systems and community gardens', *Local Environment*, 16(6), pp. 509–522. doi:10.1080/13549839.2011.569537.

Tzoulas, K. *et al.* (2007) 'Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review', *Landscape and Urban Planning*, 81(3), pp. 167–178. doi:10.1016/j.landurbplan.2007.02.001.

U.S. Census Bureau (2010a) *2010 Urban Area FAQs*. Available at: https://www.census.gov/programs-surveys/geography/about/faq/2010-urban-area-faq.html (Accessed: 20 December 2019).

U.S. Census Bureau (2010b) *Special Release - Census Blocks with Population and Housing Counts*. Available at: https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.2010.html (Accessed: 20 December 2019).

U.S. Congress (2018) *Agriculture Improvement Act of 2018*. USA. Available at: https://www.congress.gov/bill/115th-congress/house-bill/2.

U.S. Department of Energy (2016a) *CHP Technology Factsheet Series: Gas Turbines*. Available at: https://www.energy.gov/sites/prod/files/2016/09/f33/CHP-Gas Turbine.pdf.

U.S. Department of Energy (2016b) *CHP Technology Factsheet Series: Reciprocating Engines*. doi:10.1097/01.mat.0000169120.48527.87.

U.S. Department of Energy (2022) *About CHP*. Available at: https://betterbuildingssolutioncenter.energy.gov/chp/about (Accessed: 1 August 2022).

U.S. Department of the Interior (2018) *U.S. Geological Survey National Boundary Dataset*. Available at: https://catalog.data.gov/dataset/usgs-national-boundary-dataset-nbd-downloadable-data-collectionbc141 (Accessed: 20 December 2019).

U.S. EPA (2006) *Solid waste management and greenhouse gases: A life-cycle assessment of emissions and sinks*. 3rd edn. United States Environmental Protection Agency.

UGA Extension (2017) *Commercial Tomato Production Handbook. Bulletin 1312*. Athens, GA, USA: University of Georgia Extension. Available at: https://extension.uga.edu/publications/detail.html?number=B1312&title=Commercial Tomato Production Handbook.

UK Environment Agency (2011) *Think manures: A guide to manure management*. Bristol, UK.

Available at: https://www.nutrientmanagement.org/assets/12029.

UK Environment Agency (2019) *The state of the environment: soil*. London, UK. Available at: https://www.gov.uk/government/publications/state-of-the-environment/summary-state-of-the-environment-soil.

UK Environment Agency (2021) *Nitrate Vulnerable Zones (NVZ) 2021 Designations, Datasets*. Available at: https://www.data.gov.uk/dataset/77ffd32c-13db-4d83-a1f8-044c5397bc34/nitrate-vulnerable-zones-nvz-2021-designations (Accessed: 28 August 2022).

UK Government (1993) *The Crop Residues (Burning) Regulations 1993*. UK: Statutory Instruments. Available at: https://www.legislation.gov.uk/uksi/1993/1366/made.

UK Government (2003) *Waste and Emissions Trading Act 2003*. Available at: https://www.legislation.gov.uk/ukpga/2003/33/part/1/chapter/1.

UK Government (2011) *The Landfill (Maximum Landfill Amount) Regulations 2011*. UK. Available at: https://www.legislation.gov.uk/uksi/2011/2299/made.

UK Government (2016) *National Flood Resilience Review*. London, UK. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/551137/national-flood-resilience-review.pdf.

UK Government (2018) *A Green Future: Our 25 Year Plan to Improve the Environment*. London, UK. Available at: https://www.gov.uk/government/publications/25-year-environment-plan.

UK Government (2020a) *Major new measures to protect people at highest risk from coronavirus*, *Press Release*. Available at: https://www.gov.uk/government/news/major-new-measures-to-protect-people-at-highest-risk-from-coronavirus (Accessed: 14 September 2022).

UK Government (2020b) *Plans to ease guidance for over 2 million shielding*, *Press Release*. Available at: https://www.gov.uk/government/news/plans-to-ease-guidance-for-over-2-million-shielding (Accessed: 17 October 2022).

UK Met Office (2021) *UK and regional series*. Available at: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-and-regional-series (Accessed: 5 January 2023).

UK Met Office (2022a) *Climate and climate change: UK and regional series*. Available at: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-and-regional-series (Accessed: 18 August 2022).

UK Met Office (2022b) *UK climate districts map*. Available at: https://www.metoffice.gov.uk/research/climate/maps-and-data/about/districts-map (Accessed: 5 January 2023).

UK Office for Civil Society (2017) *Community Life Survey 2016-17 Self-Completion Questionnaire*. London, UK. Available at: https://www.gov.uk/government/publications/2016-to-2017-community-life-survey-questionnaire.

UK Soil Observatory (2007) *Countryside survey of topsoil in Great Britain*, *Static Maps*. Available at: http://www.ukso.org/static-maps/countryside-survey-topsoil.html (Accessed: 16 August 2022).

UK Soil Observatory (2022a) *Parent Material Model*, *Maps*. Available at: http://www.ukso.org/static-maps/parent-material-model-of-great-britain.html (Accessed: 18 August 2022).

UK Soil Observatory (2022b) *UKSO Map Viewer*. Available at: https://mapapps2.bgs.ac.uk/ukso/home.html (Accessed: 2 September 2022).

UN DESA (2019) *World Urbanization Prospects: The 2018 Revision*. New York City, NY, USA. Available at: https://population.un.org/wup/Publications/Files/WUP2018-Report.pdf.

UN DESA (2022) *World Population Prospects 2022: Summary of Results*. New York City, NY, USA. Available at: https://www.un.org/development/desa/pd/sites/www.un.org.development.desa.pd/files/wpp2022_summary_of_results.pdf.

UN General Assembly (1966) *International Covenant on Economic, Social and Cultural Rights*. United Nations. Available at: https://www.ohchr.org/en/instruments-mechanisms/instruments/international-covenant-economic-social-and-cultural-rights.

Urlić, B. *et al.* (2016) 'Phosphorus-use efficiency of kale genotypes from coastal Croatia', *Journal of Plant Nutrition*, 39(3), pp. 389–398. doi:10.1080/01904167.2015.1016174.

USDA (2019a) *2019 Organic Survey, Census of Agriculture*. U.S. Department of Agriculture. Available at: https://www.nass.usda.gov/Publications/AgCensus/2017/Online_Resources/Organics/index.php.

USDA (2019b) *FoodData Central: Kale, raw*, *United States Department of Agriculture*. Available at: https://fdc.nal.usda.gov/fdc-app.html#/food-details/168421/nutrients (Accessed: 10 May 2022).

USDA (2019c) *FoodData Central: Tomatoes, red, ripe, raw, year round average*, *United States Department of Agriculture*. Available at: https://fdc.nal.usda.gov/fdc-app.html#/food-details/170457/nutrients (Accessed: 10 May 2022).

USDA (2019d) *Fruit and Vegetable Availability, 2017, food availability data series from the Food Availability (Per Capita) Data System*. U.S. Department of Agriculture, Economic Research Service. Available at: https://www.ers.usda.gov/data-products/food-availability-per-capita-data-system/.

USDA (2019e) *USDA Organic*, *United States Department of Agriculture*. Available at: https://www.usda.gov/topics/organic (Accessed: 20 December 2019).

USDA (2022) *Urban Agriculture and Innovative Production Grants*, *United States Department of Agriculture*. Available at: https://www.usda.gov/topics/urban/grants (Accessed: 20 December 2022).

USDA ERS (2012) *U.S. Household Food Security Survey Module: Six-Item Short Form*. Washington, DC, USA: United States Department of Agriculture, Economic Research Service. Available at: https://www.ers.usda.gov/topics/food-nutrition-assistance/food-security-in-the-us/survey-tools/.

USDA NASS (2022) *2021 State Agriculture Overview: Georgia*. United States Department of Agriculture, National Agricultural Statistics Service. Available at: https://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=GEORGIA.

USDA NRCS (2022) *Soil Tech Note 23A- Carbon:Nitrogen Ratio (C:N)*. Champaign, IL, USA: United States Department of Agriculture, Natural Resources Conservation Service. Available at: https://www.nrcs.usda.gov/conservation-basics/conservation-by-state/illinois/soil-tech-note-23a-carbonnitrogen-ratio-cn.

USDA NRSC (2011) *Carbon to Nitrogen Ratios in Cropping Systems*. Greensboro, NC, USA. Available at: https://www.hamiltonswcd.org/uploads/3/7/2/3/37236909/nrcs_carbon_nitrogen.pdf.

USDA Press (2022) *USDA Advances Food System Transformation with $43 Million for Urban Agriculture and Innovative Production, Adds New Urban County Committees*, *United States Department of Agriculture*. Available at: https://www.usda.gov/media/press-releases/2022/06/03/usda-advances-food-system-transformation-43-million-urban (Accessed: 20 December 2022).

Vaarst, M. *et al.* (2018) 'Exploring the concept of agroecological food systems in a city-region context', *Agroecology and Sustainable Food Systems*, 42(6), pp. 686–711. doi:10.1080/21683565.2017.1365321.

Vanhaute, E. (2011) 'From famine to food crisis: What history can teach us about local and global subsistence crises', *Journal of Peasant Studies*, 38(1), pp. 47–65. doi:10.1080/03066150.2010.538580.

Vári, Á. *et al.* (2022) 'Disentangling the ecosystem service "flood regulation": Mechanisms and relevant ecosystem condition characteristics', *Ambio*, 51(8), pp. 1855–1870. doi:10.1007/s13280-022-01708-0.

Velickov, S. *et al.* (2014) *Study on soil and water in a changing environment*. Luxembourg: European Commission, Directorate-General for Environment. Available at: https://data.europa.eu/doi/10.2779/20608.

Velthof, G.L. *et al.* (2009) 'Integrated Assessment of Nitrogen Losses from Agriculture in EU-27 using MITERRA-EUROPE', *Journal of Environmental Quality*, 38(2), pp. 402–417. doi:10.2134/jeq2008.0108.

Venkat, K. (2012) 'Comparison of Twelve Organic and Conventional Farming Systems: A Life Cycle Greenhouse Gas Emissions Perspective', *Journal of Sustainable Agriculture*, 36(6), pp. 620–649. doi:10.1080/10440046.2012.672378.

Venugopal, R. (2015) 'Neoliberalism as concept', *Economy and Society*, 44(2), pp. 165–187. doi:10.1080/03085147.2015.1013356.

Verdoliva, S.G. *et al.* (2021) 'Controlled comparisons between soil and hydroponic systems reveal increased water use efficiency and higher lycopene and β-carotene contents in hydroponically grown tomatoes', *Scientia Horticulturae*, 279(109896). doi:10.1016/j.scienta.2021.109896.

Vermeulen, P.C.M. and Van Der Lans, C.J.M. (2011) 'Combined heat and power (CHP) as a possible method for reduction of the CO2 footprint of organic greenhouse horticulture', *Acta Horticulturae*, 915, pp. 61–68. doi:10.17660/ActaHortic.2011.915.7.

Vermeulen, S.J., Campbell, B. and Ingram, J.S. (2012) 'Climate Change and Food Systems', *Annual Review of Environment and Resources*, 37, pp. 195–222. doi:10.1146/annurev-environ-020411-130608.

Verpy, H., Smith, C. and Riecks, M. (2003) 'Attitudes and behaviors of food donors and perceived needs and wants of food shelf clients', *Journal of Nutrition Education and Behavior*, 35(1), pp. 6–15. doi:10.1016/S1499-4046(06)60321-7.

Victor, C.R. and Yang, K. (2012) 'The prevalence of loneliness among adults: A case study of the United Kingdom', *Journal of Psychology: Interdisciplinary and Applied*, 146(1–2), pp. 85–104. doi:10.1080/00223980.2011.613875.

Violante, A. *et al.* (2010) 'Mobility and bioavailability of heavy metals and metalloids in soil environments', *Journal of soil science and plant nutrition*, 10(3). doi:10.4067/S0718-95162010000100005.

Vithanage, M. *et al.* (2021) 'Compost as a carrier for microplastics and plastic-bound toxic metals into agroecosystems', *Current Opinion in Environmental Science & Health*, 24(100297). doi:10.1016/j.coesh.2021.100297.

Voluntary Action Sheffield (2020) *Sheffield Charities call on food security for all residents during COVID-19 crisis*. Available at: https://www.vas.org.uk/sheffield-charities-call-on-food-security-for-all-residents-during-covid-19-crisis/ (Accessed: 14 September 2022).

Vukina, T. and Zheng, X. (2015) 'The Broiler Industry: Competition and Policy Challenges', *Choices*, 30(2), pp. 1–6.

Wakefield, J.R.H., Bowe, M. and Kellezi, B. (2022) 'Who helps and why? A longitudinal exploration of volunteer role identity, between-group closeness, and community identification as predictors of coordinated helping during the COVID-19 pandemic', *British Journal of Social Psychology*, 61(3), pp. 907–923. doi:10.1111/bjso.12523.

Wakefield, S. *et al.* (2007) 'Growing urban health: Community gardening in South-East Toronto', *Health Promotion International*, 22(2), pp. 92–101. doi:10.1093/heapro/dam001.

Wallander, S. *et al.* (2021) *Cover Crop Trends, Programs, and Practices in the United States. Economic Information Bulletin No. EIB-222*. United States Department of Agriculture, Economic Research Service. Available at: https://www.ers.usda.gov/publications/pub-details/?pubid=100550.

Walmart (2022) *Walmart Environmental, Social and Governance Summary Report: FY 2022*. Available at: https://corporate.walmart.com/esgreport/media-library/document/walmart-fy2022-esg-summary/_proxyDocument?id=00000182-21ec-d591-afe2-2bfcb4df0000.

Wang, Y. *et al.* (2017) 'Responses of denitrifying bacterial communities to short-term waterlogging of soils', *Scientific Reports*, 7(1), p. 803. doi:10.1038/s41598-017-00953-8.

Wang, Y. *et al.* (2019) 'Estimating soil nitrate leaching of nitrogen fertilizer from global meta-analysis', *Science of the Total Environment*, 657, pp. 96–102. doi:10.1016/j.scitotenv.2018.12.029.

Ward, M. *et al.* (2018) 'Drinking Water Nitrate and Human Health: An Updated Review', *International Journal of Environmental Research and Public Health*, 15(7), p. 1557. doi:10.3390/ijerph15071557.

Warshawsky, D. and Vos, R. (2019) 'Governing at Scale: Successful Local Food Initiatives in the World's Cities', *Sustainability*, 11(24), p. 7226. doi:10.3390/su11247226.

Watson, C.A., Bengtsson, H., *et al.* (2002) 'A review of farm-scale nutrient budgets for organic farms as a tool for management of soil fertility', *Soil Use and Management*, 18(3), pp. 264–273. doi:10.1079/sum2002127.

Watson, C.A., Atkinson, D., *et al.* (2002) 'Managing soil fertility in organic farming systems', *Soil Use and Management*, 18, pp. 239–247. doi:10.1111/j.1475-2743.2002.tb00265.x.

Watts, C.W. and Dexter, A.R. (1997) 'The influence of organic matter in reducing the destabilization of soil by simulated tillage', *Soil and Tillage Research*, 42, pp. 253–275. doi:10.1016/S0167-1987(97)00009-3.

Watts, G. (2020) 'COVID-19 and the digital divide in the UK', *The Lancet Digital Health*, 2(8), pp. e395–e396. doi:10.1016/S2589-7500(20)30169-2.

Webb, J. *et al.* (2013) 'Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK?', *International Journal of Life Cycle Assessment*, 18(7), pp. 1325–1343. doi:10.1007/s11367-013-0576-2.

Weber, C.L. and Matthews, H.S. (2008) 'Food-miles and the relative climate impacts of food choices in the United States', *Environmental Science and Technology*, 42(10), pp. 3508–3513. doi:10.1021/es702969f.

Webster, P. and Neal, K. (2022) 'The "cost of living crisis"', *Journal of Public Health*, 44(3), pp. 475–476. doi:10.1093/pubmed/fdac080.

Weithmann, N. *et al.* (2018) 'Organic fertilizer as a vehicle for the entry of microplastic into the environment', *Science Advances*, 4(4). doi:10.1126/sciadv.aap8060.

Wentworth, J. (2020) *Effects of COVID-19 on the food supply system, UK Parliamentary Office of Science and Technology*. Available at: https://post.parliament.uk/effects-of-covid-19-on-the-food-supply-system/ (Accessed: 18 August 2020).

Wentworth, J. and Tresise, M. (2022) *Restoring agricultural soils: POSTnote 662*. London, UK: UK Parliamentary Office of Science and Technology. Available at: https://post.parliament.uk/research-briefings/post-pn-0662/.

Wentworth, J. and Zu Ermgassen, S. (2020) *Natural Mitigation of Flood Risk: POSTnote 623*. London, UK: UK Parliamentary Office of Science and Technology. Available at: https://post.parliament.uk/research-briefings/post-pn-0623/.

van der Werf, H.M.G., Knudsen, M.T. and Cederberg, C. (2020) 'Towards better representation of organic agriculture in life cycle assessment', *Nature Sustainability*, 3(6), pp. 419–425. doi:10.1038/s41893-020-0489-6.

Wezel, A. *et al.* (2009) 'Agroecology as a science, a movement and a practice. A review', *Agronomy for Sustainable Development*, 29, pp. 503–515. doi:10.1051/agro/2009004.

Wezel, A. *et al.* (2014) 'Agroecological practices for sustainable agriculture. A review', *Agronomy for Sustainable Development*, 34(1), pp. 1–20. doi:10.1007/s13593-013-0180-7.

Wheeler, A. (2020) *COVID-19 UK Veg Box Report*. London, UK: The Food Foundation. Available at: https://foodfoundation.org.uk/publication/covid-19-uk-veg-box-scheme-report.

Wheeler, T. and von Braun, J. (2013) 'Climate Change Impacts on Global Food Security', *Science*, 341(6145), pp. 508–513. doi:10.1126/science.1239402.

White, C. (2021) 'Our worst fears about the impact of Universal Credit cut are coming true', *BMJ*, 375(n2846). doi:10.1136/bmj.n2846.

White, E. (2012) *A life cycle assessment of a standard Irish composting process and agricultural use of compost*. Dublin, Ireland: rx3. Available at: http://www.cre.ie/web/wp-content/uploads/2010/12/Compost-Life-Cycle.pdf.

White, R. (2013) 'Land Theft as Rural Eco-Crime', *International Journal of Rural Criminology*, 1(2), pp. 203–217. doi:10.18061/1811/53698.

White, R.G. and Van Der Boor, C. (2020) 'Impact of the COVID-19 pandemic and initial period of lockdown on the mental health and well-being of adults in the UK', *BJPsych Open*, 6(5), pp. 1–4. doi:10.1192/bjo.2020.79.

Whittman, H. (2011) 'Food sovereignty. A new rights framework for food and nature?', *Environment and Society: Advances in Research*, 2, pp. 87–105.

Wholesome Wave (2022) *Georgia Fresh for Less, Programs*. Available at: https://www.wholesomewavegeorgia.org/georgiafreshforless (Accessed: 15 December 2022).

Wielemaker, R. *et al.* (2019) 'Fertile cities: Nutrient management practices in urban agriculture', *Science of The Total Environment*, 668, pp. 1277–1288. doi:10.1016/j.scitotenv.2019.02.424.

Wilhelm, J.A. and Smith, R.G. (2018) 'Ecosystem services and land sparing potential of urban and peri-urban agriculture: A review', *Renewable Agriculture and Food Systems*, 33(5), pp. 481–494. doi:10.1017/s1742170517000205.

Williams, A. *et al.* (2016) 'Contested space: The contradictory political dynamics of food banking in the UK', *Environment and Planning A: Economy and Space*, 48(11), pp. 2291–2316. doi:10.1177/0308518X16658292.

Williams, A., Goodwin, M. and Cloke, P. (2014) 'Neoliberalism, Big Society, and progressive localism', *Environment and Planning A*, 46(12), pp. 2798–2815. doi:10.1068/a130119p.

Williams, A.G. *et al.* (2009) *Comparative life-cycle assessment of food commodities procured for UK consumption through a diversity of supply chains*. Cranfield, UK: Cranfield University and Defra. Available at: https://randd.defra.gov.uk/ProjectDetails?ProjectID=15001.

Williams, A.G., Audsley, E. and Sandars, D.L. (2006) *Determining the environmental burdens and resource use in the production of agricultural and horticultural commoditites*. Bedford, UK: Cranfield University and Defra. Available at:

https://randd.defra.gov.uk/ProjectDetails?ProjectID=11442.

de Willigen, P. (2000) *An analysis of the calculation of leaching and denitrification losses as practised in the NUTMON approach*. Wageningen, Netherlands: Plant Research International, Wageningen Environmental Research. Available at: https://edepot.wur.nl/46382.

Windfuhr, M. and Jonsén, J. (2005) *Food sovereignty: towards democracy in localized food systems*. Rugby, UK: ITDG Publishing.

Windham, J.S. (2007) 'Putting Your Money Where Your Mouth Is: Perverse Food Subsidies, Social Responsibility & America's 2007 Farm Bill', *Environmental Law and Policy Journal, UC Davis School of Law*, 31(1).

Withers, P.J.A. *et al.* (2009) 'Characterization of Phosphorus Sources in Rural Watersheds', *Journal of Environmental Quality*, 38(5), pp. 1998–2011. doi:10.2134/jeq2008.0096.

Withers, P.J.A. *et al.* (2014) 'Agriculture and eutrophication: Where do we go from here?', *Sustainability (Switzerland)*, 6(9), pp. 5853–5875. doi:10.3390/su6095853.

Withers, P.J.A. and Haygarth, P.M. (2007) 'Agriculture, phosphorus and eutrophication: a European perspective', *Soil Use and Management*, 23(s1), pp. 1–4. doi:10.1111/j.1475-2743.2007.00116.x.

Wolfensberger, U. and Dinkel, F. (1997) *Beurteilung nachwachsender Rohstoffe in der Schweiz in den Jahren 1993-1996*. Bern, Switzerland: FAT und Carbotech.

Wood, C. (2012) *Destination unknown: Summer 2012*. London: Demos. Available at: https://www.demos.co.uk/files/Destination_Unknown_Summer_2012_-_web.pdf?1340294386.

Woodhouse, P. (2010) 'Beyond industrial agriculture? Some questions about farm size, productivity and sustainability', *Journal of Agrarian Change*, 10(3), pp. 437–453. doi:10.1111/j.1471-0366.2010.00278.x.

Work and Pensions Committee (2020a) *DWP's response to the coronavirus outbreak: First Report of Session 2019–21*. London, UK: House of Commons. Available at: https://publications.parliament.uk/pa/cm5801/cmselect/cmworpen/178/17814.htm.

Work and Pensions Committee (2020b) *Universal Credit: the wait for a first payment*. London, UK: House of Commons. Available at: https://committees.parliament.uk/publications/3069/documents/28787/default/.

Wortman, S.E. and Lovell, S.T. (2013) 'Environmental Challenges Threatening the Growth of Urban Agriculture in the United States', *Journal of Environmental Quality*, 42(5), pp. 1283–1294. doi:10.2134/jeq2013.01.0031.

WRAP (2020) *AD and Composting Industry Market Survey Report*. Banbury, UK: UK Waste and Resources Action Programme. Available at: https://wrap.org.uk/resources/report/anaerobic-digestion-and-composting-latest-industry-survey-report-new-summaries.

Wrenn, M. V. and Waller, W. (2017) 'Care and the Neoliberal Individual', *Journal of Economic Issues*, 51(2), pp. 495–502. doi:10.1080/00213624.2017.1321438.

Wyatt, B.M., Arnall, D.B. and Ochsner, T.E. (2019) *Nutrient Loss and Water Quality*. Stillwater, OK, USA: Oklahoma Cooperative Extension Service. Available at: https://extension.okstate.edu/fact-sheets/nutrient-loss-and-water-quality.html.

Wyer, K.E. *et al.* (2022) 'Ammonia emissions from agriculture and their contribution to fine particulate matter: A review of implications for human health', *Journal of Environmental Management*, 323(116285). doi:10.1016/j.jenvman.2022.116285.

Xu, D. *et al.* (2020) 'Modelling long-term impacts of fertilization and liming on soil acidification at Rothamsted experimental station', *Science of The Total Environment*, 713, p. 136249.

doi:10.1016/j.scitotenv.2019.136249.

Yaffe-Bellany, D. and Corkery, M. (2020) ‘Dumped Milk, Smashed Eggs, Plowed Vegetables: Food Waste of the Pandemic’, *The New York Times*, April. Available at: https://www.nytimes.com/2020/04/11/business/coronavirus-destroying-food.html.

Yancy, C.W. (2020) ‘COVID-19 and African Americans’, *JAMA*, 323(19), p. 1891. doi:10.1001/jama.2020.6548.

Yang, J.-L. and Zhang, G.-L. (2011) ‘Water infiltration in urban soils and its effects on the quantity and quality of runoff’, *Journal of Soils and Sediments*, 11(5), pp. 751–761. doi:10.1007/s11368-011-0356-1.

Yelboğa, M.N.M. (2020) ‘LCA analysis of grafted tomato seedling production in Turkey’, *Sustainability (Switzerland)*, 12(25). doi:10.3390/SU12010025.

Yoder, N. and Davis, J.G. (2020) ‘Organic fertilizer comparison on growth and nutrient content of three kale cultivars’, *HortTechnology*, 30(2), pp. 176–184. doi:10.21273/HORTTECH04483-19.

Yoshihara, T. *et al.* (2014) ‘A precise/short-interval measurement of nitrous oxide emission from a rockwool tomato culture’, *Environmental Control in Biology*, 52(3), pp. 137–147. doi:10.2525/ecb.52.137.

Yu, P. *et al.* (2018) ‘Selecting the minimum data set and quantitative soil quality indexing of alkaline soils under different land uses in northeastern China’, *Science of The Total Environment*, 616–617, pp. 564–571. doi:10.1016/j.scitotenv.2017.10.301.

Yue, Q. *et al.* (2017) ‘Mitigating greenhouse gas emissions in agriculture: From farm production to food consumption’, *Journal of Cleaner Production*, 149, pp. 1011–1019. doi:10.1016/j.jclepro.2017.02.172.

Zafiriou, P. *et al.* (2012) ‘Analysis of energy flow and greenhouse gas emissions in organic, integrated and conventional cultivation of white asparagus by PCA and HCA: Cases in Greece’, *Journal of Cleaner Production*, 29–30, pp. 20–27. doi:10.1016/j.jclepro.2012.01.040.

van der Zee, B. (2017) ‘UK is 30-40 years away from “eradication of soil fertility”, warns Gove’, *The Guardian*, 24 October. Available at: https://www.theguardian.com/environment/2017/oct/24/uk-30-40-years-away-eradication-soil-fertility-warns-michael-gove.

Zezza, A. and Tasciotti, L. (2010) ‘Urban agriculture, poverty, and food security: Empirical evidence from a sample of developing countries’, *Food Policy*, 35, pp. 265–273. doi:10.1016/j.foodpol.2010.04.007.

Zhang, W. *et al.* (2007) ‘Ecosystem services and dis-services to agriculture’, *Ecological Economics*, 64, pp. 253–260. doi:10.1016/j.ecolecon.2007.02.024.

Zhang, Yuting *et al.* (2016) ‘Long-term tobacco plantation induces soil acidification and soil base cation loss’, pp. 5442–5450. doi:10.1007/s11356-015-5673-2.

Zipfel, T. *et al.* (2015) *Our lives: Challenging attitudes to poverty in 2015*. Available at: https://www.tuc.org.uk/research-analysis/reports/our-lives-challenging-attitudes-poverty-2015.

Ziss, E. *et al.* (2021) ‘Exploring the Potential Risk of Heavy Metal Pollution of Edible Cultivated Plants in Urban Gardening Contexts Using a Citizen Science Approach in the Project “Heavy Metal City-Zen”’, *Sustainability*, 13(15), p. 8626. doi:10.3390/su13158626.

Zurek, M. *et al.* (2018) ‘Assessing sustainable food and nutrition security of the EU food system-an integrated approach’, *Sustainability (Switzerland)*, 10(11). doi:10.3390/su10114271.

Zurek, M. *et al.* (2022) ‘Food System Resilience: Concepts, Issues, and Challenges’, *Annual Review of Environment and Resources*, 47(1). doi:10.1146/annurev-environ-112320-050744.

Zwolak, A. *et al.* (2019) ‘Sources of Soil Pollution by Heavy Metals and Their Accumulation in Vegetables: a Review’, *Water, Air, and Soil Pollution*, 230(7). doi:10.1007/s11270-019-4221-y.

Appendices

This Appendix provides supporting information for the three major studies of this thesis, including: the lifecycle assessment study (supporting material in Appendix Sections A-D); the soil health assessment (Appendix Sections E-H); and the qualitative study about food insecurity (Appendix Section: I).

A. UK conventional tomato lifecycle inventory

This appendix section provides further detail on the lifecycle inventory (LCI) used for UK conventional tomato production (farms: R-C-NG and R-C-CHP). In contrast to other farm LCIs, this LCI was based on many more assumptions from literature data, as the grower could only share limited information due to the data-protective nature of the controlled environment agriculture industry.

A.1 UK conventional seedling production

The LCI for seedling phase is based on the output of the total number of seedlings transplanted on the farm (550,000), but considering the resources needed for all seedlings started (550,000).

The nursery used by the UK tomato grower produces seedlings in coco coir blocks, grown using hydroponic methods in heated glasshouses for 12 weeks. The nursery operates recirculating, or closed, hydroponic systems, so that nutrient solution run-off is recirculated back to the crops, which reduces total fertiliser and water use by 50% and 30%, respectively (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018). The nursery uses combined heat and power (CHP) for heat and electricity production. It is assumed that natural gas is used for the CHP system in the Netherlands, as this is the fuel source generally used (Karlsson, 2011; Röös and Karlsson, 2013). For the lifecycle option evaluating tomato production without CHP, it will still be assumed that natural gas is used for heating. Finally, the nursery mainly employs biocontrol and biological products for pest and disease protection.

Table 68 provides an overview of the resource and material flows for tomato seedling production within the UK conventional tomato lifecycle, with amounts provided per seedling transplanted. Information is provided based on the production of one tomato seedling over a 12-week period, including resources needed for the 10% over-sow rate. Basic information about what materials and resources are used were provided by the grower or detailed on the nursery’s website; however, specific amounts of material and resource flows were mostly estimated using secondary sources (literature or LCA databases). Materials are listed based on the amounts used per transplanted seedling, and these figures are not allocated based on time of use or lifetime of use. Allocated material flows have been previously provided within the main text (Table 27).

Water, heat, and electricity use were estimated based on averages for amounts per seedling per week from various literature sources, either for data based in the Netherlands (Karlsson, 2011; Röös and Karlsson, 2013; Quantis and Agroscope, 2019; ecoinvent Centre, 2021) or France (ADEME, 2020). These were then scaled to account for the 12-week growing period for seedlings in this study. For the assessment of tomato cultivation using combined heat and power (CHP), this will result in surplus electricity production. It is assumed that a similar type of CHP system is used in the nursery as for the cultivation glasshouse; thus, surplus electricity produced by CHP is calculated in the same way, using the same heating and electrical efficiencies (Table 70), as for tomato cultivation.

Glasshouse space use was estimated based on Karlsson (2011), who collected information from nursery growers in the Netherlands to provide a figure for the average amount of space that seedlings take up over a six-week period (as this changes week to week as the seedlings grow). This was assumed as 2.5 plant m^{-2}. Infrastructure for the glasshouse (including irrigation materials) were modelled as for tomato cultivation, based largely on AGRIBALYSE v.3.0 processes for glasshouses; see Table 73 for more detail about specific material flows for irrigation materials (ADEME, 2020).

For fertilisers, it was assumed that the same types of fertilisers were used as for tomato cultivation (Table 72), with total use for all seedlings being 0.20% of the total amount used for cultivation. There is some uncertainty within this estimation. Many glasshouse tomato LCAs and database processes do not account for the use of fertilisers during the seedling phase, even if they account for energy (Röös and Karlsson, 2013; Quantis and Agroscope, 2019; ecoinvent Centre, 2021), likely because it is a very small proportion of total fertiliser use. On the other hand, Dias et al. (2017) estimated that fertiliser use for seedlings was 3% of that used during tomato crop cultivation. However, their 3% estimate is highly uncertain, as it was based on the assumption of a nursery producer that heating requirements for tomato seedlings would be 3% of that used for tomato cultivation, and this 3% value was simply applied to fertiliser and water inputs as well.

Thus, the 0.20% estimate is much more conservative. This value is based on the amounts of fertilisers modelled in AGRIBALYSE v.3.0 processes for soilless tomato seedlings, compared to the fertiliser amounts modelled for in AGRIBALYSE processes for soilless tomato production. N, P, and K fertiliser amounts for seedling production were 0.038-0.3% (average 0.17%) of the respective amounts of fertilisers used for tomato cultivation, accounting for amounts of fertilisers applied per plant (using listed planting densities in the nurseries and glasshouses).

To further validate this value from AGRIBALYSE processes, advice from a UK hydroponic nutrient solution supplier was also taken into account. In personal communication, the supplier stated that the total electrical conductivity (often used as a proxy for total dissolved nutrients) in nutrient solution tanks for tomato seedling production was likely half the electrical conductivity used for tomato cultivation. Thus, it could be assumed that the amounts of nutrients in the water is roughly half for tomato seedlings. Nutrients supplied to tomato seedlings was then estimated based on amounts of water supplied to the seedlings, assuming they would have approximately half of the nutrients per water amount supplied to cultivated tomatoes. This rough estimation equated to 0.27% of the total amount of nutrients supplied to cultivated tomatoes being supplied to the seedlings, which is within the ranges calculated for AGRIBALYSE processes. Thus, the average of all figures was taken to provide an estimate of 0.2%. There is high uncertainty in this figure, and thus an uncertainty range is included based on values provided for different nutrients in the AGRIBALYSE processes (0.038-0.29%). This estimate is applied for all fertilisers (both macronutrients and micronutrients), as used for tomato cultivation. This fertiliser input for tomato seedlings is further reduced by 50%, since the nursery operates a closed hydroponic system, and it is estimated that closed systems achieve a fertiliser reduction of 50% (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018).

For pest and disease protection, it is assumed that the same basic items and products are employed by the nursery as done for cultivation, as both specify using biocontrol and biological protection products. Thus, the amounts of insect traps and pheromone disruption products, as used in tomato production glasshouses, have been scaled for use in the nursery and allocated to tomato seedlings based on 2.5 seedlings grown m^{-2} over a 12-week period. The amounts of biological pesticides applied are based on previous literature (ADEME, 2020; Yelboğa, 2020); however, the specific types used are assumed to be the same proportions as for tomato cultivation (see Appendix Section A.2 for further detail).

Table 68 *– Resource and material flows for UK conventional tomato seedling production*

Inputs	**Unit**	**Amount per transplanted seedling (range)**	**Source**
Water use [a]	L	3.47 (1.33-5.07)	Average from: (Quantis and Agroscope, 2019; ADEME, 2020; ecoinvent Centre, 2021)
Heating requirement [b]	kWh	1.83 (0.13-4.53)	Average from: (Karlsson, 2011; Röös and Karlsson, 2013; Quantis and Agroscope, 2019; ADEME, 2020; ecoinvent Centre, 2021)
Natural gas use (boiler)	m^3	0.188	Calculated using Equation 30
Natural gas use (total, CHP)	m^3	0.374	Calculated using Equation 30
Electricity use	kWh	0.32 (0.033-0.64)	Average from: (Karlsson, 2011; Röös and Karlsson, 2013; ADEME, 2020)
Glasshouse space use	m^2	0.44	Amount of space from (Karlsson, 2011). Infrastructure modelled with AGRIBALYSE v.3.0 processes.
Glasshouse time use	weeks	12	Primary data (grower)
Coconut coir block substrate	g	103	Primary data (supplier)
Mesh for coir block (PP)	g	2.0	Primary data (supplier)
Plastic crates for transport (PP)	g	64, assume 75 uses in lifetime [b]	Primary data (grower); lifetime is estimate.
Drip irrigation system (PE)	g	19.33 (11.0-35.9)	(Antón *et al.*, 2012; Torrellas, Antón, López, *et al.*, 2012; Bojacá, Wyckhuys and Schrevens, 2014)
Macronutrient fertilisers (N, P, K, Mg, Ca, S) [c]	mg	2,222	(ADEME, 2020) and estimation from nutrient solution supplier
Micronutrient fertilisers (B, Cu, Fe, Zn, Mn, Mo) [c]	mg	26.7	(ADEME, 2020) and estimation from nutrient solution supplier
Insect sticky traps (PE with polybutene adhesive)	mg	3.22	Primary data (grower and supplier); components based on (Lo, Wallis and Bellamy, 2019)
Pheromone disruptors	mg, ai	1.1 (0.98-1.22)	Primary data (supplier)
Biological insecticides	mg, ai	0.150 (0.032-0.269)	Average from: (ADEME, 2020; Yelboğa, 2020)
Biological fungicides	mg, ai	0.91 (0.542-1.28)	Average from: (ADEME, 2020; Yelboğa, 2020)
Biocontrol (predatory insects) [d]	number	9.08	(Williams, Audsley and Sandars, 2006)
Transport of seedlings from nursery to farm, by road	km, one-way	553	Primary data (grower)
Transport of seedlings from nursery to farm, by train	km, one-way	54	Primary data (grower)
Transport of seedlings from nursery to farm, by boat	km, one-way	11	Primary data (grower).
Outputs			
Seedlings, transported to farm	number	1	Primary data (grower)
Seedling waste (equivalent to % over-sow)	number	0.1	Primary data (grower)
Surplus electricity produced (CHP) [e]	kWh	1.42	Calculated using Equation 31 and Equation 32

[a] This is the base (unadjusted) water requirement for tomato seedling production. This value is further adjusted in the model to account for a 30% water reduction due to use of closed hydroponic systems (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018).

[b] The heating requirement is the amount of heat output needed for the seedlings. This is estimated based on heat usage as given in the literature sources listed, and back-calculated to obtain the heat requirement by using the boiler and glasshouse heating efficiencies as provided in Table 70.

[c] It is assumed that the same fertilisers are used as in tomato cultivation, in the same proportions; for more information on specific fertilisers used, see Table 72. The amount of fertilisers needed for tomato seedlings, as listed in this table, is the base requirement. In this LCA model, this value has been adjusted to account for the fact that the nursery operates a recirculating (closed) hydroponic system, assuming that closed systems use 50% less fertilisers (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018).

[d] Burdens for biocontrol products are based on the approximation used by (Williams, Audsley, and Sandars 2006), which includes just transport to the farm (10 kg*km by van) and 2.78 kWh energy input, per m^2 glasshouse over the whole year.

[e] For the conventional tomato lifecycle assuming combined heat and power use, this will result in surplus electricity being produced. Surplus electricity allocation is dealt with using either energy allocation (in Scenarios 1 and 2) or system expansion (in Scenario 3).

A.2 UK conventional tomato cultivation

Table 69 provides an overview of the main resource flows for the cultivation phase of UK conventional tomato production, assembled using both primary data from the grower and supplier and secondary data from LCA databases and LCA studies on hydroponic tomato production. Flows are provided per gross glasshouse cultivation area, in m^2. Specific flows are discussed in more detail in the following sections. The outputs of the cultivation phase include the harvested crop (estimated based on yield amounts), harvest waste, crop green waste, and surplus electricity (for the CHP system only).

Table 69 – *Resource flows for cultivation in UK conventional tomato production*

Inputs	**Unit**	**Amount m^{-2} (range)**	**Source**
Water use [a]	L	1,345 (681-1,772)	Average based on (Williams, Audsley and Sandars, 2006). Uncertainty range from (Boulard *et al.*, 2011; Antón *et al.*, 2012; Almeida *et al.*, 2014; Dias *et al.*, 2017; Grasselly, Trédan and Colomb, 2018; Quantis and Agroscope, 2019; ADEME, 2020)
Heating requirement [b]	kWh	327 (218-513)	(Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011; Vermeulen and Van Der Lans, 2011; Antón *et al.*, 2012; ADEME, 2020)
Natural gas use (boiler)	m^3	33.48	Calculated using Equation 30
Natural gas use (total, CHP)	m^3	66.59	Calculated using Equation 30
Electricity use	kWh	11.2 (9.00-18.3)	(Williams, Audsley and Sandars, 2006; Blonk *et al.*, 2010; Boulard *et al.*, 2011; Karlsson, 2011; Antón *et al.*, 2012; Almeida *et al.*, 2014; Grasselly, Trédan and Colomb, 2018; Nemecek *et al.*, 2019; Quantis and Agroscope, 2019; ADEME, 2020; ecoinvent Centre, 2021)
Glasshouse [c]	m^2	1	Modelled with AGRIBALYSE v.3.0 processes (ADEME, 2020).
Macronutrient fertilisers (N, P, K, Mg, Ca, S) [d]	kg	2.25	Literature sources - see Table 71 and Table 72.
Micronutrient fertilisers (B, Cu, Fe, Zn, Mn, Mo) [d]	kg	0.028	Literature sources - see Table 71 and Table 72.
Pheromone disruption products	g, ai	0.0108 (0.0096-0.012)	Primary data (supplier).
Biological insecticides	g, ai	0.59 (0.09-2.03)	(Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011; Karlsson, 2011; Antón *et al.*, 2012)
Biological fungicides	g, ai	0.15 (0.14-0.70)	(Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011; Karlsson, 2011; Antón *et al.*, 2012)
Biocontrol (predatory insects)	number	88.7	(Williams, Audsley and Sandars, 2006)
Bumblebee colonies	number	0.0058	(Williams, Audsley and Sandars, 2006)
Transport of crop, with tractor and trailer (round-trip)	km	3.22	Primary data (grower).
Outputs			
Harvested crop	kg	34.3	UK industry data, as provided by grower.
Harvest waste	kg	0.19	(Hueso-Kortekaas, Romero and González-Felipe, 2021)
Surplus electricity (CHP) [f]	kWh	300	Calculated using Equation 31 and Equation 32

[a] This is the base water requirement for hydroponic tomato production in the UK. In the LCA model, this number has been adjusted based on the grower's assumption that 50% of hydroponic systems are closed (recirculate the water and nutrient solution), and that closed systems use 30% less water (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018). Thus, the amount after adjusting for the use of recirculating systems is: 1,143.5 (579-1,506) L m^{-2}.
[b] Heating requirement is the amount of output heat required for the tomato crops, not the energy value of the natural gas input. Note that this heat requirement is modelled in two cases: as conventional heating through natural gas and electricity through the Great British national grid (ecoinvent Centre, 2021), or using co-production of heat and electricity through combined heat and power (CHP).
[c] This inventory is modelled per m^2 glasshouse production. The material components in the glasshouse structure has been based on the AGRIBALYSE v.3.1 process for a glasshouse with a metal frame (ADEME, 2020).
[d] This is the sum of all fertilisers (whole compounds, not individual nutrients). The specific nutrient requirements and fertilisers used are presented in Table 71 and Table 72, respectively. Note that this represents the base amount of fertilisers required for hydroponic tomato production in the UK. In the LCA model, this number has been adjusted based on the grower's assumption that 50% of hydroponic systems are closed (recirculate the water and nutrient solution), and that closed systems use 50% less fertilisers (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018).
[f] For the conventional tomato lifecycle assuming combined heat and power use, this will result in surplus electricity being produced. This is calculated using Equation 31, Equation 32, and the thermal and electrical efficiencies provided in Table 70. Surplus electricity allocation is dealt with using either energy allocation (in Scenarios 1 and 2) or system expansion (in Scenario 3), with methods outlined in pg. 152.

Yield

Yields have been defined based on averaged industry data in 2019 for conventional UK glasshouse tomato production in 2019. This is based on a weighted average of yields for all tomato varieties produced, based on areas of production across the country. This includes the classic 'salad' tomatoes as well as speciality varieties such as cocktail, cherry, plum (mini, midi, and max), and beef, marketed both as loose and 'on the vine.' This industry data was provided by the grower through the British Tomato Growers' Association. The weighted average yield used is thus 34.3 kg harvested crop per m^2 glasshouse area.

Waste

Harvest waste is estimated as 0.55% of the total harvest, based on a figure provided by another LCA study on hydroponic tomato production (Hueso-Kortekaas, Romero and González-Felipe, 2021). This amount seems reasonable, as the grower specified that there is a 5% non-marketable yield, which includes waste and also tomatoes sold through secondary channels (not to supermarkets). It is assumed that the majority of this non-marketable yield sold through the secondary sales channels (artisan business, often processed into sauces), so this is not counted as waste. However, the grower designated that a small portion of tomatoes fall on the floor and thus are wasted, and this amount is thus based on the 0.55% harvest waste figure.

Energy use

Heating is required during the colder parts of the year for conventional glasshouse tomato production in the UK. Electricity, in much smaller amounts, is also used for lighting, irrigation, and general operational needs. Specific data on heating and energy use could not be provided by the growers in this study; thus, this has been approximated with literature data. Values used are averaged from studies with the most relevant or similar production systems to the system of interest, which is characterised by year-round, hydroponic production of tomatoes in heated glasshouses in colder climates like the UK. In addition to the last study on UK tomato production (Williams, Audsley and Sandars, 2006), geographies of the included studies are: France (Boulard *et al.*, 2011; Grasselly, Trédan and Colomb, 2018; ADEME, 2020), the Netherlands (Blonk *et al.*, 2010; Karlsson, 2011; Antón *et al.*, 2012; Röös and Karlsson, 2013; Nemecek *et al.*, 2019; ecoinvent Centre, 2021), and Sweden (Karlsson, 2011; Röös and Karlsson, 2013). Uncertainty ranges are derived from the minimum and maximum of values provided in these studies.

As this study investigates the use of two energy systems, it is imperative to know the heating requirement of the glasshouse (necessary energy output) so that the actual input energy can be calculated for the two systems (conventional boiler and CHP). Most of the relevant LCA studies previously listed only provide information on the amount of gas or energy used to heat the glasshouse. Thus, this has been back-calculated to estimate the heat requirement based on thermal efficiencies either provided within the studies themselves, or if not available, using those as provided within this study (Table 70) as a proxy. Then, the energy input or volume of gas required to satisfy this requirement can be calculated, as described in the following section.

System efficiencies and natural gas inputs

The amount of heat and electricity produced from a CHP unit depend on its thermal and electrical efficiencies (i.e., how much natural gas is converted to power in the forms of heat and electricity, respectively), which in turn depend on the type of system and size of generator used. The amount of natural gas needed to satisfy the heating requirements for tomato production can be calculated using these CHP efficiencies, as well as the heating efficiency of the glasshouse (related to how much heat is utilised or lost within the glasshouse).

For this LCA, the CHP system modelled is based on that used by the farm of interest; this same system is also assumed for the nursery glasshouse. The UK conventional tomato grower in this LCA has two energy centres for its 27.5 ha of glasshouse production, each which operate two 5.605 MW reciprocating engines (internal combustion engines). These engines use natural gas as the fuel source. Thermal and electrical efficiencies for each engine is rated at 49.8% and 44.84% respectively, with a 5% error tolerance; according to the energy supplier, this likely results in a 46.5% thermal efficiency and 42.5% electrical efficiency, giving a total efficiency of 89% (88-90% range). When comparing electrical and thermal efficiency ratings between sources, it must be noted whether the efficiency is being reported based on the gross calorific value (GCV) of the fuel, also called the higher heating value (HHV), or the net calorific value (NCV), also called the lower heating value (LHV). The efficiencies provided for the CHP system in this study are in terms of GCV, so all values used in this section have been re-calculated, when necessary, to be in terms of GCV, based on a gross:net calorific value ratio of 1.109 for natural gas (BEIS, 2021b).

Comparing the thermal and electrical efficiencies of the engines used at this farm to those used in other LCA sources, it is seen that these efficiencies are slightly higher than many, but overall comparable. Blonk et al. (2010) and Antón et al. (2012) estimate a total CHP efficiency of 81%, with 45% thermal efficiency and 36% electrical efficiency, based on CHP systems used in horticultural greenhouses in the Netherlands. Vermeulen and Van Der Lans (2011) estimate 90% total efficiency, with 50% thermal and 40% electric, also for glasshouses in the Netherlands. Williams, Audsley, and Sandars (2006), in analysing hydroponic tomato production in the UK, estimated a 43% thermal efficiency and 40% electrical efficiency for CHP systems used in these glasshouses. No distinction is made for the type of CHP system used in these studies, although practitioners are encouraged to use site specific information. UK BEIS (2021b) estimates that reciprocating engines for CHP have an approximate 47.5% thermal efficiency and 31% electrical efficiency, while the U.S. DOE (2016b) estimates a 38.6% thermal efficiency and 40.9% electrical efficiency for a 3.3 MWe reciprocating gas engine. The efficiency values provided by these sources were used to produce the uncertainty range used within this LCA, as shown in Table 70.

As for CHP systems, boilers also have a rated heating efficiency. The boiler heating efficiency value used in this study is 92.5%, as based on that used by Williams, Audsley, and Sandars (2006) in a prior study on UK conventional tomato production (information provided directly by first author). Heating efficiency in the glasshouse depends on the glasshouse structure and its insulative potential, among other factors. For this study, a value of 96% will be assumed, based on that achieved by heated glasshouses in the Netherlands (Blonk *et al.*, 2010). The efficiency values thus provided have been summarised in Table 70.

Table 70 *– Thermal and electrical efficiencies for energy systems in tomato glasshouse production*

Efficiency rating	**Amount**	**Uncertainty range**
Conventional boiler heating efficiency [a]	92.5%	80-95%
CHP electrical efficiency [b]	42.5%	25-45%
CHP thermal efficiency [b]	46.5%	37-50%
CHP total efficiency (%) [b]	89%	62-95%
Glasshouse heating efficiency (%) [c]	96%	n/a

[a] Efficiency for conventional boiler heating using natural gas input. This is based on that used by Williams, Audsley, and Sandars (2006), with specific information provided by the study authors. Uncertainty range estimated based on (BEIS, 2021a).
[b] Efficiencies based on information provided by engine supplier and energy company, using the gross calorific value of natural gas. Uncertainty range based on minimum and maximum values used throughout other studies on glasshouse production with CHP or use of CHP reciprocating engines (Williams, Audsley and Sandars, 2006; Blonk *et al.*, 2010; Antón *et al.*, 2012; U.S. Department of Energy, 2016a; BEIS, 2021b).
[c] Based on (Blonk *et al.*, 2010).

These efficiency values determine the amount of natural gas needed to satisfy heating requirements, both for the conventional heating system and the CHP unit. Heating requirements for the tomato glasshouses in this study exceed electrical requirements; thus, the natural gas input for heat is first considered for the CHP system. The natural gas input per area of glasshouse is calculated using Equation 30.

$$NG_{input} = \frac{heat_{req}}{\left(t_{eff} * h_{eff}\right)} * \frac{1}{NG_{GCV}}$$

Equation 30 – Calculation of natural gas input to CHP unit for UK conventional tomato production

Where,

- NG_{input} = the amount of natural gas input to the CHP unit needed to satisfy the tomato heating requirement, in [m^3 natural gas m^{-2} glasshouse]
- $heat_{req}$ = the amount of heating energy required for UK conventional tomato production, in [kWh m^{-2} glasshouse]. This is provided in the relevant LCIs for tomato seedlings and cultivation (Table 25 and Table 31, respectively).
- t_{eff} = the thermal efficiency of the boiler or CHP unit as provided in Table 70, as [%] (expressed in decimal form).
- h_{eff} = the heating efficiency of the glasshouse as provided in Table 70, as [%] (expressed in decimal form).NG_{GCV} = the gross calorific value (GCV) of natural gas, in [kWh m^{-3}]. In this study, the value of 11 kWh m^{-3} is used, based on UK BEIS (2021b) guidance.

For the CHP system, additional consideration is required because the natural gas burned to provide heating will also produce electricity. The amount of electricity generated from the CHP unit whilst satisfying the heat demand can be calculated using Equation 31.

$$E_{produced} = NG_{input} * NG_{GCV} * e_{eff}$$

Equation 31 – Electricity produced from CHP unit while satisfying heating demand for UK conventional tomato production

Where,

- $E_{produced}$ = the total electricity produced by the CHP unit based on the amount of natural gas used to satisfy the glasshouse heating requirements, in [kWh m^{-2}].
- NG_{input} = the amount of natural gas input to the CHP unit needed to satisfy the tomato heating requirement, in [m^3 natural gas m^{-2} glasshouse]
- NG_{GCV} = the gross calorific value (GCV) of natural gas, in [kWh m^{-3}]. In this study, the value of 11 kWh m^{-3} is used, based on UK BEIS (2021b) guidance.
- e_{eff} = the electrical efficiency of the CHP unit as provided in Table 70, as [%] (expressed in decimal form).

Some of this electricity will be used to satisfy the electricity requirements, as designated in the relevant lifecycle inventories. However, as the electricity requirement is much lower than the heating requirements for the glasshouse, excess electricity will be produced. This surplus electricity can be calculated using Equation 32.

$$E_{surplus} = E_{produced} - E_{req}$$

Equation 32 – Surplus electricity from CHP unit for UK conventional tomato production

Where,

- $E_{surplus}$ = the surplus electricity produced by the CHP unit, in [kWh m^{-2}]
- $E_{produced}$ = the total electricity produced by the CHP unit, in [kWh m^{-2}]
- E_{req} = the electricity used within the glasshouse [kWh m^{-2}]. This is provided in the relevant LCIs for tomato seedlings and cultivation (Table 25 and Table 31, respectively).

Utilising Equation 30 through Equation 32 and considering the electricity requirement for tomato production as given in Table 69, this gives an approximate surplus electricity value of 300 kWh m^{-2}. This value is higher than surplus electricity values from CHP as provided by similar glasshouse tomato LCA studies based in the Netherlands and Italy, which have ranged from 165-203 kWh m^{-2} (Antón *et al.*, 2012; Röös and Karlsson, 2013; Almeida *et al.*, 2014), although lower than the value of 467 kWh m^{-2} used within the last UK conventional tomato LCA study (Williams, Audsley and Sandars, 2006). Thus, it is assumed that the surplus electricity calculated here is within the range of reasonable values.

Infrastructure

The glasshouse structure is modelled based on glasshouse with a metal frame using AGRIBALYSE v.3.0 processes (ADEME, 2020). The AGRIBALYSE glasshouse model is based on information from glasshouse suppliers in the North of France and specifically on systems used for the soilless cultivation of tomatoes. This model assumes the use of the glasshouse over a year, and attributes the lifetime of each material input. The glasshouse structure as a whole is expected to have a 25-year minimum lifetime. The model has been updated to include only the components which are relevant to UK hydroponic tomato production and which are within the system boundaries of this LCA. The included glasshouse model includes the following components: overall structure (e.g., glass, steel, aluminium), electrical components, heating and cooling systems (e.g., pipe network), fertigation / hydroponic system (e.g., gutters, pipes, taps, endcaps), fertiliser tanks, thermal screens, CO_2 injection system, storage facilities and floor trolley systems. It also accounts for energy use by glasshouse suppliers and by the growers during construction. Other equipment accounted for elsewhere includes the drip irrigation system (microtubes and drippers), trellis system, plastic mulch, and energy system (CHP turbine).

Pesticides

The grower in this study specified that pesticide use was minimal, as care is taken to control the environment to prevent disease and pest outbreaks (e.g., limited visitors entering, high hygiene standards). Further, only insecticides and fungicides are used; herbicides are not necessary since all growing is done in a soilless media and the ground of the glasshouse is covered in plastic mulch. Pest control is mainly done through use of sticky traps, pheromone disruption products and biological control (predatory insects). Biological pesticides (bio-pesticides) are also used only when needed.

Amounts of pesticides used were not specified by the grower, and thus this has been approximated with literature data from studies with similar growing conditions (soilless cultivation in heated glasshouses, using integrated pest management) (Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011; Karlsson, 2011; Antón *et al.*, 2012). It was assumed that pesticide use had likely decreased since last assessment of UK hydroponic tomato production (Williams, Audsley and Sandars, 2006); thus, averages of the four studies were used to approximate pesticide amounts, except when the average was higher than that of the UK study; in this case (for fungicides only), the amount in the UK study was used. However, uncertainty ranges based on the minimum and maximum values between all four studies were included. Sulphur is also commonly used as a preventative fungicide, but this is not included in the fungicide amount, since the use of sulphur is included within the macronutrient fertiliser category.

Specific active ingredients used will be estimated based on information provided by the grower (when possible) and common biopesticides used for hydroponic tomato crops. Specifically, active ingredients of pyrethrins, *Bacillus thuringiensis*, and abamectin will be used to approximate bio-insecticide use (assuming these are used in equal amounts). The grower specified use of pyrethrin, while the other active ingredients were selected based on bioinsecticides listed in other hydroponic tomato LCAs (Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011). For bio-fungicides, strains of bacterial and fungal species such as *Bacillus subtilis*, *Bacillus amyloliquefaciens*, and *Trichoderma asperellum*, or extracts of yeast (*Saccharomyces cerevisiae*), are commonly used, based on market products. For this study, the active ingredient of *Bacillus subtilis* will be used to approximate emissions from bio-fungicide use, as other types were not modelled within the modelling software.

Little information is available on biological control using predatory insects, and also the use of pollinators, although this practice is common in glasshouse production and is employed by the grower. Burdens will be approximated based on the approximation used by Williams, Audsley and Sandars (2006), which includes just transport to the farm (10 kg*km by van) and

2.78 kWh energy input, per m^2 glasshouse. This is based on the use of 88.7 predatory insects and 0.0058 bumblebee colonies per m^2 glasshouse (Williams, Audsley and Sandars, 2006).

Nutrient inputs

Although different literature sources evaluate different types of tomato crops (e.g., loose classic, on the vine, or specialty types), it has been specified that generally inputs do not vary based on the type of tomato, although yields do vary (Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011). Thus, it can be assumed that information on inputs given in units of kg m^{-2} glasshouse across studies should be comparable. However, Williams, Audsley and Sandars (2006) did adjust nutrient inputs based on tomato type, which will be accounted for.

The amount of major nutrients (N, P_2O_5, K_2O, CaO, MgO, and SO_3) supplied to tomato crops has been estimated based on literature data, shown in Table 71 in g m^{-2} glasshouse area. Sulphur has been included in this section as a nutrient, but it is also used as a preventative fungicide. The given amounts are averages of reported nutrients of eight different tomato lifecycle inventories across six different countries (UK, France, Netherlands, Sweden, Italy, and Canada) (Williams, Audsley and Sandars, 2006; Boulard *et al.*, 2011; Röös and Karlsson, 2013; Almeida *et al.*, 2014; Dias *et al.*, 2017; Grasselly, Trédan and Colomb, 2018; Nemecek *et al.*, 2019; ADEME, 2020). These studies were selected as they all utilise hydroponic glasshouse or greenhouse production, rockwool substrates and year-round production; in addition, they report yields in the range of 38-62 kg m^{-2} over the year, similar to UK yield ranges (25-54 kg m^{-2}). It should be noted that Williams, Audsley and Sandars (2006) provided the last in-depth LCA for UK glasshouse tomato production; the lifecycle inventory data used in this report was provided directly by the author, and it is this detailed data which is being utilised to supplement the lifecycle inventory for this study.

Boulard *et al.* (2011) and Williams, Audsley and Sandars (2006) both considered market mixes of different tomato types in France and England, respectively, adjusting for different yields. These included mainly "classic loose", "on the vine", and "specialist" varieties (such as cherry, cocktail, and plum). Boulard *et al.* (2011) specified that fertiliser inputs per area were similar across all tomato types; however, Williams, Audsley and Sandars (2006) adjusted nutrient inputs per area for each tomato type as proportions of the "classic loose" fertiliser inputs, based on the relative yields of each tomato type (with classic loose having the highest yields). Thus, the nutrient inputs for the classic loose varieties were the highest, whilst nutrient inputs for all other types were lower. To utilise the data from this study, the weighted average of the production area of each tomato type in the UK in 2019 was multiplied by the adjusted nutrient values to gain a market mix nutrient average. These values were then used in the calculations of average nutrient input across all studies, as displayed in Table 71, although the full range of values for all tomato types was used to generate maximums and minimums in the displayed uncertainty ranges.

Only Williams, Audsley and Sandars (2006) provided detail on the amounts of all micronutrients needed (B, Cu, Fe, Mn, Mo, and Zn); thus, micronutrient inputs were based only on this study. Again, the nutrient input amounts were adjusted based tomato type, and the reported values are based on the weighted average of all tomato types in the UK in 2019.

Table 71 *– Nutrient inputs for UK conventional tomato production*

Nutrient inputs	Amount (g m^{-2})	Uncertainty range
N	166 [a-g]	102-425
P_2O_5	75.5 [a-g]	0-204
K_2O	316 [a-g]	102-636
CaO	171 [a,b,f,g]	86.0-253
MgO	118 [a-c,f,g]	24.1-320
SO_3	342 [c,f,g]	73.8-742
B	0.33 [a]	0.24-0.70
Cu	0.14 [a]	0.10-0.30
Fe	2.27 [a]	1.63-4.80
Zn	0.64 [a]	0.46-1.40
Mn	0.33 [a]	0.24-0.70
Mo	0.07 [a]	0.05-0.10

[a](Williams, Audsley and Sandars, 2006), using supplementary information provided by Williams via email; [b](Boulard *et al.*, 2011) [c](Grasselly, Trédan and Colomb, 2018; ADEME, 2020); [d](Mouron *et al.*, 2017; Nemecek *et al.*, 2019; Quantis and Agroscope, 2019); [e](Röös and Karlsson, 2013); [f](Almeida *et al.*, 2014); [g](Dias *et al.*, 2017)

Specific fertilisers were then modelled to supply these nutrient amounts. Table 72 provides the fertilisers modelled and the amounts needed in g m^{-2} of glasshouse area. Fewer LCA studies have published information on specific fertiliser use; thus, fertilisers were selected based on those that did report specific fertiliser types (Almeida *et al.*, 2014; Dias *et al.*, 2017; Grasselly, Trédan and Colomb, 2018; ADEME, 2020). First, the percent of each major nutrient (N, P_2O_5, K_2O, CaO, MgO, and SO_3) supplied by each listed fertiliser, out of the total nutrient requirement reported by each study, was determined. For example, Almeida *et al.* (2014) reports that calcium nitrate supplies 65.5% and potassium nitrate supplies 34.5% of the total N requirement. These relative proportions were then averaged across the similar fertilisers reported by each study. Fertiliser amounts for this LCA were then determined based on each fertiliser's average contributions to the total nutrient requirements. Adjustments were made to ensure that the fertilisers satisfied all nutrient requirements, whilst staying within or close to the range of nutrient and fertiliser values provided in the utilised studies.

None of the utilised studies reported specific micronutrient fertilisers, so the types of micronutrient fertiliser modelled were based on the nutrient solutions components provided by the University of Florida Extension Service for hydroponic tomatoes (Hochmuth and Hochmuth, 2018).

Table 72 – *Fertiliser inputs for UK conventional tomato production*

Fertiliser inputs [a]	Amount (g m^{-2})	Uncertainty range
Calcium nitrate	515	315-1,243
Potassium nitrate	451	267-1,214
Ammonium nitrate	49.6	30.3-127
Phosphoric acid	208	0-560
Potassium sulphate	234	44.9-254
Magnesium sulphate	739	150-2,001
Calcium chloride	52.4	0-58.2
Sodium borates	1.61	1.16-3.41
Copper sulphate	0.568	0.408-1.2
Iron sulphate	22.7	16.3-48
Zinc sulphate	1.79	1.28-3.89
Manganous sulphate	1.18	0.85-2.5
Sodium molybdenate	0.167	0.121-0.25

[a] Fertiliser types were chosen based on (Almeida *et al.*, 2014; Dias *et al.*, 2017; Grasselly, Trédan and Colomb, 2018; Hochmuth and Hochmuth, 2018; ADEME, 2020)

Run-off recycling

It is estimated that closed hydroponic systems for tomato cultivation lead to a 30% reduction in water use and 50% reduction in fertiliser use (Martinez and Morard, 2000; Grasselly, Trédan and Colomb, 2018). It is assumed that 50% of the hydroponic systems used within the case study farm are open and 50% are closed, based on information provided by the grower. Thus, the fertiliser and water reductions for closed systems are considered for 50% of production on the case study farm. However, this has also been explored within the sensitivity analysis to see how much this influences final results, considering cases of 100% open and 100% closed systems.

Material Flows

Materials used for UK conventional tomato production are provided in Table 73 based on the amount used (in grams) per gross glasshouse cultivation area (in m^2). Material amounts have been allocated to tomatoes based on time of use. Lifetimes of materials are also provided, but note that these have not been applied to the material amounts listed. Materials listed are primarily based on those defined by the grower. Specific suppliers (as designated by the grower) were contacted to provide additional information regarding amounts used, material types, and weights of products, where this was not provided by the grower. Where grower or supplier information could not be obtained, secondary data from literature sources were used to estimate amounts.

***Table 73** – Material flows for UK conventional tomato production*

Item	Material	Amount (g m^{-2})	Lifetime (years)	Source
Coco coir substrate	Coconut fibre	582	1	Primary data (grower and supplier)
Rockwool substrate	Stonewool	4.09	1	Primary data (grower and supplier)
Substrate plastic	LDPE	12.3	1	Primary data (supplier).
Plastic mulch	LDPE	68.3	2	Primary data; film density from (Williams, Audsley and Sandars, 2006)
Drip irrigation system (flexible pipes, drippers, microtubes, and pickaxes)	PE	43.8 (23-79)	5	(Antón *et al.*, 2012; Torrellas, Antón, López, *et al.*, 2012; Bojacá, Wyckhuys and Schrevens, 2014)
Pond liner, for reservoir	Butyl	1.6	15	Primary data (grower and supplier)
Twine for trellis	Jute	881	1	Primary data (grower); density from (Williams, Audsley and Sandars, 2006)
Hooks for trellis	Steel	232	5	Primary data (grower and supplier). Lifetime from (Williams, Audsley and Sandars, 2006)
Wire for trellis	Steel	126 (113-139)	30	(Torrellas, Antón, López, *et al.*, 2012; Bojacá, Wyckhuys and Schrevens, 2014)
Insect sticky traps	PE with polybutene adhesive	0.0316	1 time use	Primary data (grower and supplier); components based on (Lo, Wallis and Bellamy, 2019)
Harvest crates[a]	HDPE	248	15	Primary data (grower).

[a] Plastic crates are used for harvesting tomatoes into, but also for storing the crop. So have attributed half of the total weight of crates used to the cultivation stage, and half will also be attributed to the processing stage. This is done similarly for all farms.

A.3 UK conventional tomato processing and storage

Tomatoes are harvested into plastic crates, then immediately placed in an on-farm cool store (12°C) for 24 hours (maximum). They are then transported to a separate packhouse site where packaging takes place. It is assumed that the main energy usage for the on-farm storage facility (75 m^2) is for the cool storage, which houses tomatoes all year long. Thus, the energy use for this has been estimated based on PEFCR guidance for cool storage of crops in distribution centres, which estimates 40 kWh m^{-3} cool storage space (European Commission, 2018). This simply assumes electricity from the grid, since this generates is an extremely small value per m^2 of production for tomatoes. Similarly, electricity supply from the national grid is assumed for the main packhouse operation which takes place at a different site.

Electricity use for packhouse operations and cool storage in the packhouse, as well as propane use for packhouse operations, has been estimated based on figures from other UK vegetable farmers in this study. Propane is used mainly for forklifts. The figure has been obtained by applying the total propane usage per all crops harvested on each farm, then applying this to the amount of harvested tomatoes for UK conventional production. The electricity figure, however, is based on the annual electricity usage for packhouse operations given by these farmers, divided by the area of their packhouses to obtain a figure per m^2 packhouse, since it is assumed electricity usage should be proportional to packhouse area. The figure used for electricity is likely a conservative estimate to the packhouse operations for the tomato farm, since the other vegetable farmers in this study grow more leafy greens which require washing and chopping. Annual values are then allocated to tomatoes based on the glasshouse area of the tomato site of interest divided by the total glasshouse area this packhouse serves, which is 59%. It is assumed that mostly tomatoes are grown across these sites, likely with a smaller amount of other fruiting vegetable crops, so this is an approximate estimate for allocating storage space in the packhouse, since the farmer could not provide figures for mass allocation.

__Table 74__ – Main resource flows for processing and storage of UK conventional tomatoes

Resource flows	**Unit**	**Amount m^{-2} glasshouse area (range)**	**Source**
Inputs			
Electricity use, on-farm cool storage (24 hour)	kWh	0.016	(European Commission, 2018)
Electricity use, packhouse operations and cool storage [a]	kWh	7.34 (6.46-8.22)	Estimation based figures provided by other UK vegetable farmers in study
Propane use [a]	kg	0.17 (0.09-0.25)	Estimation based figures provided by other UK vegetable farmers in study
On-farm storage	m^2	0.00032	Primary data (grower)
Packhouse	m^2	0.016	Primary data (grower)
Transport to packhouse	kg*km	1,920	Primary data (grower)
Outputs			
Sellable crop	kg	33.4	Primary data (grower)
Processing waste	kg	0.86	Primary data (grower)

[a] The figure for total electricity usage in a packhouse with cold storage has been estimated based on figures provided by other UK conventional vegetable farmers in this study, which have similar sized packhouses and cool stores.

B. Averaged LCIA results for farm categories

This appendix section provides LCIA results for all 18 midpoint impact categories generated by the ReCiPe 2016 Hierarchist impact method, with a subset of these results being discussed in Section 3.1.1. These are presented as average values for each allocation scenario (1-3) and farm category, including: urban organic (U-O), peri-urban organic (PU-O), rural organic (R-O), and rural conventional (R-C), also considering two cases for UK conventional tomato production (R-C-NG for heating with natural gas and R-C-CHP for heating via CHP). In particular, results are presented in Table 75 for U.S. kale lifecycle; Table 76 for UK kale lifecycles; Table 77 for U.S. tomato lifecycles; and Table 78 for UK tomato lifecycles.

Table 75 *- Lifecycle impact assessment results for U.S. kale production, as an average of scenarios and farm categories*

		Scenario 1				Scenario 2				Scenario 3			
Impact category	**Unit, per kg kale**	U-O (n=1)	PU-O (n=4)	R-O (n=3)	R-C (n=1)	U-O (n=1)	PU-O (n=4)	R-O (n=3)	R-C (n=1)	U-O (n=1)	PU-O (n=4)	R-O (n=3)	R-C (n=1)
Global warming	kg CO_2 eq	5.94	3.93	2.83	1.10	1.97	3.23	2.75	1.10	-15.8	-0.95	1.97	1.10
Stratospheric ozone depletion	kg CFC-11 eq	8.3E-05	2.0E-05	1.1E-05	1.6E-05	2.5E-05	9.2E-06	1.0E-05	1.6E-05	8.1E-05	2.0E-05	1.1E-05	1.6E-05
Ionizing radiation	kBq Co-60 eq	0.546	0.232	0.341	0.041	0.260	0.207	0.330	0.041	0.477	0.217	0.339	0.041
Ozone formation, Human health	kg NOx eq	1.3E-02	7.4E-03	6.1E-03	3.5E-03	9.3E-03	6.6E-03	5.9E-03	3.5E-03	4.7E-03	5.5E-03	5.7E-03	3.5E-03
Fine particulate matter formation	kg PM2.5 eq	1.1E-02	5.6E-03	4.7E-03	2.1E-03	3.3E-03	4.0E-03	4.4E-03	2.1E-03	8.7E-03	5.0E-03	4.6E-03	2.1E-03
Ozone formation, Terrestrial ecosystems	kg NOx eq	1.4E-02	7.8E-03	6.3E-03	3.6E-03	9.4E-03	6.9E-03	6.1E-03	3.6E-03	4.6E-03	5.7E-03	5.9E-03	3.6E-03
Terrestrial acidification	kg SO_2 eq	5.3E-02	1.8E-02	1.0E-02	9.6E-03	6.8E-03	8.6E-03	9.0E-03	9.6E-03	4.7E-02	1.7E-02	1.0E-02	9.6E-03
Freshwater eutrophication	kg P eq	2.1E-03	9.4E-04	1.3E-03	2.1E-04	1.4E-03	8.7E-04	1.2E-03	2.1E-04	1.6E-03	8.3E-04	1.2E-03	2.1E-04
Marine eutrophication	kg N eq	1.0E-02	6.1E-04	1.5E-03	2.7E-03	9.8E-03	5.6E-04	1.5E-03	2.7E-03	-1.3E-02	-4.5E-03	6.0E-04	2.7E-03
Terrestrial ecotoxicity	kg 1,4-DCB	22.7	15.1	10.0	34.6	13.9	13.2	9.68	34.6	17.2	13.9	9.82	34.6
Freshwater ecotoxicity	kg 1,4-DCB	0.307	0.284	0.249	0.133	0.137	0.254	0.243	0.133	-11.60	-2.39	-0.225	0.133
Marine ecotoxicity	kg 1,4-DCB	0.405	0.368	0.317	0.129	0.180	0.328	0.309	0.129	-15.21	-3.14	-0.304	0.129
Human carcinogenic toxicity	kg 1,4-DCB	0.446	0.619	0.675	0.072	0.204	0.571	0.665	0.072	-0.030	0.512	0.656	0.072
Human non-carcinogenic toxicity	kg 1,4-DCB	14.4	5.73	3.64	2.24	10.9	5.07	3.50	2.24	-242	-51.7	-6.55	2.24
Land use	m^2a crop eq	0.566	0.609	1.34	0.320	0.423	0.582	1.33	0.32	0.465	0.587	1.33	0.320
Mineral resource scarcity	kg Cu eq	0.014	0.017	0.013	0.003	0.011	0.016	0.012	0.003	0.012	0.017	0.013	0.003
Fossil resource scarcity	kg oil eq	0.922	1.019	0.791	0.265	0.444	0.944	0.772	0.265	0.439	0.910	0.772	0.265
Water consumption	m^3	0.170	0.139	0.288	0.123	0.157	0.137	0.288	0.123	0.161	0.138	0.288	0.123

Table 76 - *Lifecycle impact assessment results for UK kale production, as an average of scenarios and farm categories*

		Scenario 1				**Scenario 2**				**Scenario 3**			
Impact category	**Unit, per kg kale**	U-O (n=1)	PU-O (n=4)	R-O (n=2)	R-C (n=5)	U-O (n=1)	PU-O (n=4)	R-O (n=2)	R-C (n=5)	U-O (n=1)	PU-O (n=4)	R-O (n=2)	R-C (n=5)
Global warming	kg CO_2 eq	1.59	1.65	0.60	1.37	1.59	0.68	0.58	1.37	1.59	-3.21	-0.42	1.37
Stratospheric ozone depletion	kg CFC-11 eq	1.5E-05	3.5E-05	8.5E-06	1.5E-05	1.5E-05	1.3E-05	1.0E-05	1.5E-05	1.5E-05	2.8E-05	7.9E-06	1.5E-05
Ionizing radiation	kBq Co-60 eq	0.043	0.062	0.040	0.150	0.043	0.022	0.021	0.150	0.043	0.053	0.039	0.150
Ozone formation, Human health	kg NOx eq	1.0E-02	4.0E-03	2.9E-03	6.0E-03	1.0E-02	3.5E-03	2.7E-03	6.0E-03	1.0E-02	1.6E-03	2.5E-03	6.0E-03
Fine particulate matter formation	kg PM2.5 eq	3.2E-03	2.8E-03	1.5E-03	2.1E-03	3.2E-03	1.7E-03	1.2E-03	2.1E-03	3.2E-03	2.1E-03	1.4E-03	2.1E-03
Ozone formation, Terrestrial ecosystems	kg NOx eq	1.0E-02	4.1E-03	2.9E-03	6.1E-03	1.0E-02	3.6E-03	2.7E-03	6.1E-03	1.0E-02	1.6E-03	2.5E-03	6.1E-03
Terrestrial acidification	kg SO_2 eq	6.6E-03	1.7E-02	6.6E-03	5.8E-03	6.6E-03	8.5E-03	4.7E-03	5.8E-03	6.6E-03	1.5E-02	6.3E-03	5.8E-03
Freshwater eutrophication	kg P eq	2.7E-04	4.9E-04	2.1E-04	4.0E-04	2.7E-04	4.7E-04	2.0E-04	4.0E-04	2.7E-04	2.5E-04	1.6E-04	4.0E-04
Marine eutrophication	kg N eq	7.4E-03	3.1E-03	1.8E-03	3.1E-03	7.4E-03	3.1E-03	1.8E-03	3.1E-03	7.4E-03	8.4E-04	1.3E-03	3.1E-03
Terrestrial ecotoxicity	kg 1,4-DCB	3.42	2.63	1.80	8.88	3.42	1.82	1.41	8.88	3.42	-0.108	1.52	8.88
Freshwater ecotoxicity	kg 1,4-DCB	0.071	0.065	0.050	0.098	0.071	0.046	0.041	0.098	0.1	-2.45	-0.506	0.098
Marine ecotoxicity	kg 1,4-DCB	0.094	0.084	0.064	0.126	0.094	0.060	0.052	0.126	0.1	-3.20	-0.662	0.126
Human carcinogenic toxicity	kg 1,4-DCB	0.387	0.201	0.146	0.174	0.387	0.174	0.134	0.174	0.387	-0.021	0.099	0.174
Human non-carcinogenic toxicity	kg 1,4-DCB	1.54	29.7	12.1	1.40	1.54	29.4	11.9	1.40	1.54	-18.9	1.32	1.40
Land use	m^2a crop eq	4.44	1.26	1.72	1.20	4.44	1.24	1.72	1.20	4.44	1.24	1.72	1.20
Mineral resource scarcity	kg Cu eq	0.011	0.003	0.003	0.008	0.011	0.003	0.003	0.008	0.011	0.002	0.003	0.008
Fossil resource scarcity	kg oil eq	0.419	0.195	0.151	0.380	0.419	0.145	0.127	0.380	0.419	0.113	0.138	0.380
Water consumption	m^3	0.010	0.009	0.037	0.027	0.010	0.007	0.036	0.027	0.010	0.002	0.035	0.027

Table 77 *- Lifecycle impact assessment results for U.S. tomato production, as an average of scenarios and farm categories*

		Scenario 1				Scenario 2				Scenario 3			
Impact category	**Unit, per kg tomato**	U-O (n=1)	PU-O (n=3)	R-O (n=3)	R-C (n=2)	U-O (n=1)	PU-O (n=3)	R-O (n=3)	R-C (n=2)	U-O (n=1)	PU-O (n=3)	R-O (n=3)	R-C (n=2)
Global warming	kg CO_2 eq	2.25	1.75	4.17	0.35	0.89	1.47	3.58	0.35	-5.31	-0.38	0.72	0.35
Stratospheric ozone depletion	kg CFC-11 eq	2.8E-05	1.1E-05	1.7E-05	3.2E-06	8.2E-06	6.1E-06	9.0E-06	3.2E-06	2.7E-05	1.1E-05	1.7E-05	3.2E-06
Ionizing radiation	kBq Co-60 eq	0.212	0.087	0.387	0.014	0.112	0.085	0.342	0.014	0.188	0.080	0.376	0.014
Ozone formation, Human health	kg NOx eq	5.2E-03	4.1E-03	7.2E-03	1.5E-03	3.8E-03	3.8E-03	6.5E-03	1.5E-03	2.2E-03	3.2E-03	5.8E-03	1.5E-03
Fine particulate matter formation	kg PM2.5 eq	4.3E-03	2.9E-03	5.9E-03	6.7E-04	1.5E-03	2.3E-03	4.7E-03	6.7E-04	3.3E-03	2.6E-03	5.5E-03	6.7E-04
Ozone formation, Terrestrial ecosystems	kg NOx eq	5.3E-03	4.2E-03	7.4E-03	1.5E-03	3.8E-03	3.9E-03	6.8E-03	1.5E-03	2.2E-03	3.3E-03	6.0E-03	1.5E-03
Terrestrial acidification	kg SO_2 eq	1.9E-02	1.0E-02	1.8E-02	2.1E-03	3.1E-03	5.9E-03	1.1E-02	2.1E-03	1.7E-02	9.6E-03	1.7E-02	2.1E-03
Freshwater eutrophication	kg P eq	7.3E-04	5.0E-04	1.2E-03	1.9E-04	4.9E-04	4.9E-04	1.1E-03	1.9E-04	5.6E-04	4.5E-04	1.1E-03	1.9E-04
Marine eutrophication	kg N eq	3.2E-03	2.2E-03	6.6E-04	6.47E-04	3.1E-03	2.2E-03	6.2E-04	6.5E-04	-4.7E-03	-1.4E-05	-2.9E-03	6.5E-04
Terrestrial ecotoxicity	kg 1,4-DCB	8.47	7.61	13.1	5.64	5.44	6.80	11.8	5.64	6.59	7.08	12.26	5.64
Freshwater ecotoxicity	kg 1,4-DCB	0.13	0.13	0.27	0.04	0.07	0.12	0.24	0.04	-4.00	-1.03	-1.62	0.04
Marine ecotoxicity	kg 1,4-DCB	0.17	0.18	0.34	0.09	0.10	0.16	0.31	0.09	-5.25	-1.34	-2.13	0.09
Human carcinogenic toxicity	kg 1,4-DCB	0.34	0.73	0.75	0.02	0.25	0.71	0.72	0.02	0.17	0.68	0.68	0.02
Human non-carcinogenic toxicity	kg 1,4-DCB	5.41	6.52	4.97	0.50	4.20	6.27	4.42	0.50	-83.5	-18.4	-35.6	0.50
Land use	m^2a crop eq	0.296	0.980	0.599	0.200	0.246	0.969	0.577	0.200	0.261	0.970	0.583	0.200
Mineral resource scarcity	kg Cu eq	0.006	0.012	0.017	0.002	0.004	0.012	0.016	0.002	0.005	0.012	0.017	0.002
Fossil resource scarcity	kg oil eq	0.417	0.413	1.083	0.109	0.251	0.388	1.008	0.109	0.249	0.366	1.007	0.109
Water consumption	m^3	0.054	0.086	0.196	0.069	0.050	0.085	0.194	0.069	0.051	0.085	0.195	0.069

Table 78 *– Lifecycle impact assessment results for UK tomato production, as an average of scenarios and farm categories*

		Scenario 1					Scenario 2					Scenario 3				
Impact category	**Unit, per kg tomato**	U-O (n=2)	PU-O (n=4)	R-O (n=2)	R-C NG (n=1)	R-C CHP (n=1)	U-O (n=2)	PU-O (n=4)	R-O (n=2)	R-C NG (n=1)	R-C CHP (n=1)	U-O (n=2)	PU-O (n=4)	R-O (n=2)	R-C NG (n=1)	R-C CHP (n=1)
Global warming	kg CO_2 eq	1.75	1.22	0.30	4.19	3.73	0.56	0.56	0.26	4.24	3.78	-4.46	-2.38	-0.07	4.19	3.18
Stratospheric ozone depletion	kg CFC-11 eq	2.9E-05	2.2E-05	2.4E-06	2.7E-06	2.1E-06	7.9E-06	5.1E-06	2.3E-06	4.2E-06	3.6E-06	2.5E-05	2.0E-05	2.2E-06	2.7E-06	1.4E-06
Ionizing radiation	kBq Co-60 eq	0.202	0.050	0.023	0.237	0.142	0.102	0.043	0.016	0.237	0.142	0.191	0.043	0.023	0.237	-1.646
Ozone formation, Human health	kg NOx eq	4.1E-03	2.5E-03	1.4E-03	4.9E-03	4.1E-03	3.0E-03	2.5E-03	1.3E-03	4.9E-03	4.1E-03	9.3E-04	7.0E-04	1.2E-03	4.9E-03	-2.6E-05
Fine particulate matter formation	kg PM2.5 eq	4.0E-03	1.8E-03	8.5E-04	2.4E-03	2.0E-03	1.9E-03	1.3E-03	7.4E-04	2.4E-03	2.0E-03	3.2E-03	1.3E-03	8.2E-04	2.4E-03	6.1E-05
Ozone formation, Terrestrial ecosystems	kg NOx eq	4.2E-03	2.6E-03	1.4E-03	5.1E-03	4.6E-03	3.0E-03	2.5E-03	1.3E-03	5.1E-03	4.6E-03	9.7E-04	7.4E-04	1.3E-03	5.1E-03	7.8E-04
Terrestrial acidification	kg SO_2 eq	2.6E-02	1.0E-02	3.2E-03	6.3E-03	4.8E-03	0.010	0.006	0.002	0.007	0.005	0.024	0.009	0.003	0.006	-0.001
Freshwater eutrophication	kg P eq	5.7E-04	3.9E-04	1.3E-04	8.0E-04	7.5E-04	5.1E-04	3.8E-04	1.2E-04	8.0E-04	7.5E-04	2.5E-04	2.0E-04	1.1E-04	8.0E-04	2.8E-04
Marine eutrophication	kg N eq	1.0E-03	5.0E-04	9.4E-05	6.2E-04	6.1E-04	9.5E-04	5.0E-04	9.1E-05	6.2E-04	6.1E-04	-2.0E-03	-1.2E-03	-8.6E-05	6.2E-04	5.5E-04
Terrestrial ecotoxicity	kg 1,4-DCB	3.34	2.24	1.38	26.0	25.7	1.28	2.09	1.24	26.0	25.7	-0.26	0.15	1.28	26.0	23.9
Freshwater ecotoxicity	kg 1,4-DCB	0.09	0.04	0.06	0.28	0.26	0.04	0.04	0.06	0.28	0.26	-3.22	-1.87	-0.14	0.28	0.23
Marine ecotoxicity	kg 1,4-DCB	0.11	0.06	0.08	0.35	0.34	0.05	0.05	0.07	0.35	0.34	-4.20	-2.44	-0.19	0.35	0.30
Human carcinogenic toxicity	kg 1,4-DCB	0.45	0.30	0.12	0.67	0.63	0.38	0.29	0.12	0.67	0.63	0.15	0.13	0.11	0.67	0.54
Human non-carcinogenic toxicity	kg 1,4-DCB	30.3	18.8	4.66	3.64	3.42	29.6	18.8	4.61	3.64	3.42	-33.7	-18.3	0.78	3.64	2.10
Land use	m^2a crop eq	0.6	0.29	0.35	0.14	0.13	0.6	0.28	0.35	0.15	0.13	0.6	0.28	0.35	0.14	-0.10
Mineral resource scarcity	kg Cu eq	0.005	0.004	0.002	0.021	0.020	0.004	0.004	0.002	0.021	0.020	0.003	0.003	0.002	0.021	0.017
Fossil resource scarcity	kg oil eq	0.25	0.16	0.09	1.45	1.32	0.12	0.15	0.08	1.45	1.32	0.14	0.10	0.08	1.45	1.18
Water consumption	m^3	0.071	0.124	0.024	0.051	0.050	0.067	0.124	0.024	0.051	0.050	0.062	0.119	0.024	0.051	0.038

C. Pesticides and Health: POSTbrief

From May through July 2021, I completed a fellowship at the Parliamentary Office of Science & Technology (POST), which was funded by the Institute for Food Science & Technology. The aim of this fellowship is to gain experience in science policy roles, with a particular focus on science communication for policy-makers. During this three-month fellowship, I completed a Parliamentary briefing on the subject of Pesticides & Health. This process requires rapidly acquiring specialist knowledge about often unfamiliar topics. To gain expert insight to my topic, I interviewed a total of 54 stakeholders during 31 interviews, which also included multi-stakeholder workshops that I led. This included academic experts as well as representatives from a wide range of relevant stakeholder groups, including: Defra (Department of Environment, Food, & Rural Affairs), the Health & Safety Executive, Public Health England, the Department for International Trade, the Expert Committee on Pesticides, the Expert Committee on Pesticide Residues in Food, pesticide manufacturers, food industry groups (i.e., the Food & Drink Federation and the Institute for Food Science & Technology), organic farming groups, the National Farmers' Union, and environmental and food charities (Pesticide Action Network, Food Ethics Council). The vast information provided by stakeholders and experts was distilled into a 33-page briefing, written for a Parliamentary audience, within two months. This briefing then went through both an internal and external review process, where the briefing was assessed by other staff and the department head within the Parliamentary Office of Science & Technology, as well as key stakeholders and experts; after this review and editing process, the briefing was published in September 2021.

I also worked with the University of Sheffield press team to communicate this research briefing widely through a press release. The briefing was shared in the Yorkshire Post as well as other relevant news sites, such as Farming UK. Further, I spoke about the research on BBC Radio 4 (Farming Today) and was also interviewed by POST about my fellowship experience. The briefing can be found online at the POST website: https://post.parliament.uk/research-briefings/post-pb-0043/ (Kennard and Vagnoni, 2021). A brief summary of the parliamentary briefing, as written by the primary author of this thesis and published with the briefing online, is provided below.

People can be exposed to pesticides in a variety of ways. They can be directly exposed whilst mixing or applying pesticides, either at work (often leading to the highest exposure levels, as more concentrated products are used) or at home (e.g., when using garden products). They can also be indirectly exposed to lower doses of pesticides through the environment (air, water, soil and dust) and through ingestion of pesticide residues in food and drink. Legal requirements of training and certification for professional applicators, pesticide storekeepers and people selling pesticides help ensure safe and sustainable use. Pesticides must go through a rigorous risk assessment process for approval, requiring a large suite of tests. Companies must prove no harmful effects on human health and no unacceptable effects on the environment for substances to be approved. Assessments will specifically consider the risks to more susceptible groups that could have higher exposure levels, such as children, those working with pesticides, and those living or walking near pesticide-treated areas. Once pesticides are approved, legal limits will also be set for the maximum amount of pesticide residue allowed in foods (known as maximum residue levels).

Adverse health impacts from pesticides depend on the toxicity of the substance; dose (amount) of exposure; duration and frequency of exposure; route of body entry (skin contact, ingestion, or inhalation); and personal vulnerability. Acute exposure to high doses of pesticides might result in poisoning. UK surveillance programmes track pesticide exposures and assess potential health risks from poisonings, for occupational users and from residues in food and drinks. Most cases of acute exposures reported in the UK were unintentional and resulted in little harm. These occurred mostly from exposure to non-professional products in the home. No causal relationships between pesticide exposure and chronic health impacts have yet been proven, although correlations have been seen. The research in this area is highly inconclusive, and the topic is difficult to research. As people are exposed to a wide range of chemicals every day, it is difficult quantify exact exposure levels, find control populations for comparisons, and associate a specific adverse health effect with an individual pesticide. Additionally, there are other confounding variables that can influence health, such as lifestyle factors (e.g., exercise and diet).

After EU withdrawal, the nations of Great Britain (i.e. England, Wales, and Scotland) now make independent decisions on pesticide approvals. Northern Ireland complies with EU law. The Health and Safety Executive (HSE) is the regulatory authority for the whole UK that reviews new pesticide applications, ensures safe use of pesticides and oversees monitoring and surveillance programmes. Some stakeholders are concerned that different decisions from the EU and within UK nations could bring regulatory challenges, complicate trade and potentially affect farmers' access to pesticides. Currently, the UK Government and the devolved administrations aim to minimise the risks and impacts of pesticides on human health and the environment through the National Action Plan (NAP) for the Sustainable Use of Pesticides. An updated version is due to be finalised by the end of 2021. The draft NAP aims to develop targets by 2022 to reduce the risks associated with pesticide use.

D. Farmers’ motivations for LCA study participation

All farmers that participated in the LCA study (Chapter 3) were asked about their motivations for participating. The basic sentiments of their statements were categorised, and this was used to generate a word cloud, as provided in Figure 75. Basic motivations from farmers were separated based on country, with 11 farmers in the U.S. and 14 farmers in the UK. The answers that occur most frequently are represented with larger sized text, thus allowing for the most common motivations among farmers in each country to be identified.

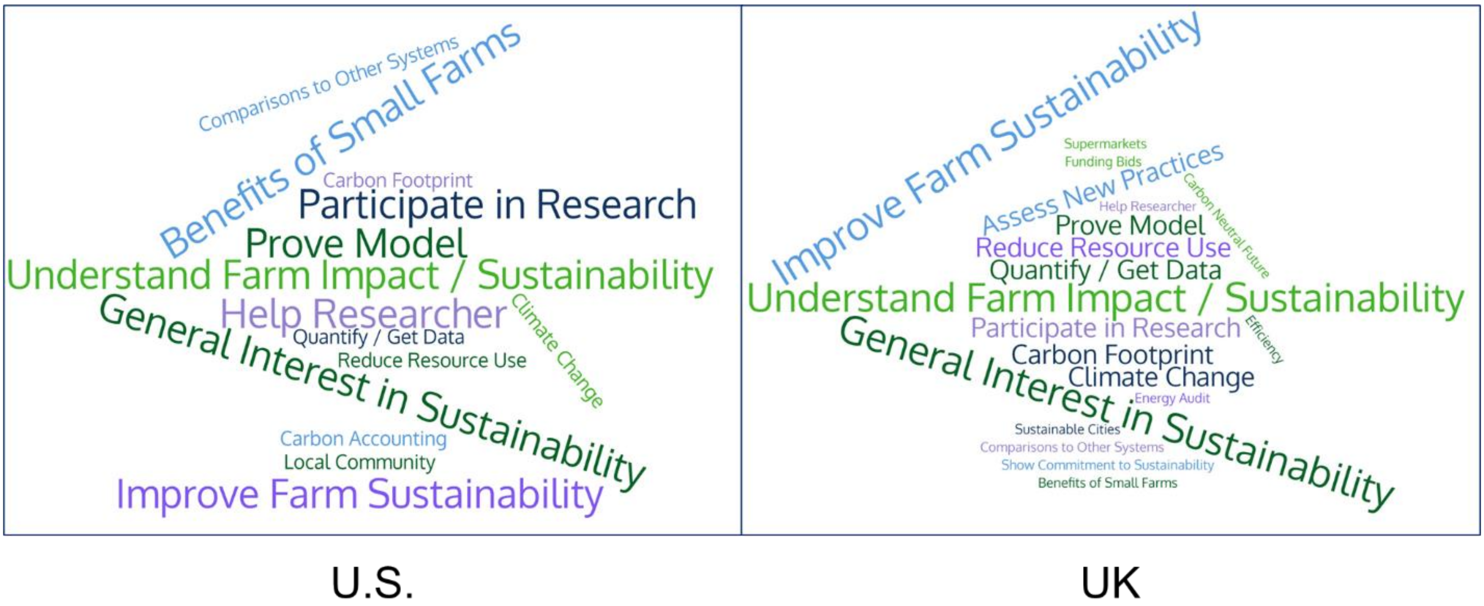

__Figure 75__ – Word cloud showing motivations of farmers to participate in the LCA study, separated based on those in the U.S. (n=11) and UK (n=14).

E. Limits of detection for ICP-MS

Table 79 displays the limits of detection (LOD) from the ICP-MS, used to evaluate elemental content of aqua regia digested soil samples, with relevant methodology for this study provided in Section 2.2.3 of this thesis. Values are displayed for the main elements presented in the results of this study (Chapter 4).

Table 79 *– Limits of detection from ICP-MS for aqua regia digested soil samples*

Element	P	As	Cd	Cu	Fe	Pb	Zn
LOD (mg kg^{-1})	<0.001	<0.001	<0.001	0.32	40.3	1.25	11.6

F. Descriptive statistics for soil parameters

Appendix Table 80 displays the mean, median, and range of values (minimum-maximum) for tested soil parameters as presented in Chapter 4 of this thesis, grouped by urbanised and rural soils. Appendix Table 81 displays the same information for soil parameters grouped by management type, which includes uncultivated, organically-managed, and conventionally-managed soils. Soil parameters listed include bulk density (BD), water-holding capacity (WHC), soil organic carbon concentration (SOC), N concentration (N), and C/N ratio, as well as concentrations of other elements including P, As, Cd, Cu, Fe, Pb, and Zn. For soil parameters listed per management type, C storage, or stock (kg C m^{-2}), is also provided for 0-10 cm, 10-20cm, and 0-20cm (total) depths. Soil parameters tested on samples from upper depths (0-10cm) are indicated by a '(u)', while those tested on samples from lower depths (10-20cm) are indicated by an '(l)'. Finally, soil parameters that had significant differences between groups (p<0.05) are indicated in boldface, based on the statistical tests outlined in main text Table 66 and Table 67 for soil parameters grouped by urbanisation and management type, respectively. Note that C storage values were not investigated for significant differences.

Table 80 – Descriptive statistics for soil parameters, grouped by urbanisation. Soil parameters resulting in significant differences (p<0.05) are indicated in bold.

		Urbanised			**Rural**		
Soil parameter	**Unit**	**Mean**	**Median**	**Range**	**Mean**	**Median**	**Range**
BD (u)	g cm^{-3}	0.90	0.93	0.61-1.22	1.03	0.99	0.63-1.74
BD (l)	g cm^{-3}	1.00	1.03	0.49-1.32	1.11	1.10	0.79-1.63
WHC (u)	%	70.1	70.6	25.6-118	64.8	62.7	48.0-91.6
WHC (l)	%	62.1	62.4	43.9-87.9	59.9	56.8	45.2-90.9
SOC (u)	mg g^{-1}	45.6	41.0	2.97-91.7	35.83	33.20	0.99-121
SOC (l)	mg g^{-1}	36.8	28.4	1.26-92.8	34.99	27.95	0.17-110
N (u)	mg g^{-1}	3.96	2.66	0.52-48.4	3.27	1.80	0.05-20.3
N (l)	mg g^{-1}	4.15	1.67	0.32-34.8	2.38	1.24	0.05-24.7
C/N (u)	Ratio	21.9	20.0	0.06-39.2	54.2	19.2	0.09-297
C/N (l)	Ratio	23.6	24.5	0.05-49.7	76.6	30.5	9.47-688
C/N (u)*	Ratio	23.1	21.1	11.7-39.2	15.8	14.4	7.96-30.8
C/N (l)*	Ratio	25.3	24.7	9.09-49.7	21.9	20.3	9.47-43.7
P (u)	mg kg^{-1}	869	739	211-1,883	809	686	232-1,695
P (l)	mg kg^{-1}	760	722	155-1,995	687	667	228-1,312
As (u)	mg kg^{-1}	13.1	10.5	5.21-59.0	21.8	8.49	1.16-146
As (l)	mg kg^{-1}	12.0	10.3	5.64-35.3	21.8	8.22	2.12-132
Cd (u)	mg kg^{-1}	0.52	0.30	0.05-5.71	0.24	0.19	0.03-0.62
Cd (l)	mg kg^{-1}	0.39	0.37	0.07-0.95	0.20	0.14	0.01-0.60
Cu (u)	mg kg^{-1}	26.1	24.3	7.65-71.2	29.5	20.9	5.86-100
Cu (l)	mg kg^{-1}	23.4	22.0	8.80-56.8	28	21	5.01-94.7
Fe (u)	mg kg^{-1}	19,600	19,087	11,497-31,804	17,014	12,782	5,492-46,188
Fe (l)	mg kg^{-1}	20,442	20,184	14,508-27,943	17,207	12,773	4,955-44,197
Pb (u)	mg kg^{-1}	141	82	36.12-782	50	39	11.1-221
Pb (l)	mg kg^{-1}	136	82	30.9-589	47	42	11.6-126
Zn (u)	mg kg^{-1}	131	107	58.4-404	140	55	<11.6-1,322
Zn (l)	mg kg^{-1}	107	105	49.0-197	117	64	12.5-591

*Excluding outliers values <1 and ≥60.

***Table 81** – Descriptive statistics for soil parameters, grouped by management type. Soil parameters resulting in significant differences (p<0.05) are indicated in bold.*

		Uncultivated			Organic			Conventional		
Soil parameter	**Unit**	**Mean**	**Median**	**Range**	**Mean**	**Median**	**Range**	**Mean**	**Median**	**Range**
BD (u)	g cm^{-3}	0.89	0.87	0.61-1.74	0.93	0.94	0.69-1.32	1.27	1.30	1.06-1.49
BD (l)	g cm^{-3}	1.01	1.01	0.49-1.42	1.02	1.03	0.69-1.45	1.28	1.31	0.92-1.63
WHC (u)	%	68.0	69.9	25.6-91.6	71.0	70.2	51.5-118	56.0	54.3	48.0-71.9
WHC (l)	%	62.4	61.0	47.7-90.9	62.6	62.4	43.9-87.9	51.9	49.6	47.5-61.7
SOC (u)	mg g^{-1}	49.1	43.3	4.75-121	40.2	40.4	0.99-91.7	18.1	21.0	5.75-33.2
SOC (l)	mg g^{-1}	37.2	27.9	1.26-110	40.2	37.3	0.17-92.8	18.9	22.3	4.46-36.7
C stock (u)	kg C m^{-2}	7.84	6.87	1.57-21.5	6.92	6.84	0.20-13.4	4.24	4.93	1.10-6.94
C stock (l)	kg C m^{-2}	6.27	4.84	0.15-21.6	7.74	6.74	1.53-19.1	4.60	5.28	0.52-10.1
C stock (0-20 cm)	kg C m^{-2}	14.1	10.5	2.36-35.4	14.7	13.6	3.08-32.5	8.83	10.4	1.61-12.4
N (u)	mg g^{-1}	3.11	2.33	0.17-20.26	4.75	3.28	0.1-48.4	1.80	1.09	0.05-7.95
N (l)	mg g^{-1}	5.47	2.04	0.05-34.81	2.02	1.92	0.16-5.16	0.72	0.34	0.11-1.75
C/N (u)	Ratio	39.4	20.2	7.96-297	28.9	19.9	0.06-270	54.3	21.7	14.4-201
C/N (l)	Ratio	58.2	24.5	0.05-688	39.7	24.7	9.47-287	44.1	31.2	13.6-99.8
C/N (u)*	Ratio	20.3	19.4	7.96-39.2	20.5	19.9	8.85-39.2	18.4	18.4	14.4-24.6
C/N (l)*	Ratio	26.02	24.3	9.08-26.0	22.50	24.4	9.47-36.4	22.39	20.3	13.6-35.0
P (u)	mg kg^{-1}	672	545	211-1,636	1082	1053	439-1,883	616	500	400-1,319
P (l)	mg kg^{-1}	549	485	179-1,393	946	892	155-1,995	590	537	402-916
As (u)	mg kg^{-1}	17.1	8.79	1.16-88.92	10.1	9.16	3.33-21.5	39.3	9.29	4.70-146
As (l)	mg kg^{-1}	16.0	9.01	2.12-91.14	10.5	10.3	3.2-22.9	37.2	10.20	5.12-132
Cd (u)	mg kg^{-1}	0.48	0.21	0.03-5.71	0.36	0.30	0.05-1.05	0.23	0.16	0.11-0.47
Cd (l)	mg kg^{-1}	0.25	0.14	0.01-0.74	0.37	0.30	0.12-0.95	0.21	0.15	0.12-0.41
Cu (u)	mg kg^{-1}	24.0	17.8	5.86-71.19	26.9	25.8	7.65-54.8	41.8	28.4	12.5-100
Cu (l)	mg kg^{-1}	20.7	17.0	5.01-62.01	25.3	24.3	8.80-56.8	39.9	27.1	14.2-94.7
Fe (u)	mg kg^{-1}	18,467	18,559	5,492-42,724	18,052	18,839	11,115-25,046	19,250	12,059	5,566-46,188
Fe (l)	mg kg^{-1}	18,792	19,245	4,955-43,407	18,844	19,108	10,312-27,943	19,673	14,217	6,322-44,197
Pb (u)	mg kg^{-1}	104	56	11.1-782	106	76	17.4-313	58	57	14.9-107
Pb (l)	mg kg^{-1}	89	60	11.6-589	111	76	15.4-288	60	63	14.7-115
Zn (u)	mg kg^{-1}	99	66	0-404	127	107	35.6-389	273	122	18.3-1,322
Zn (l)	mg kg^{-1}	86	68	12.5-320	108	105	33.0-201	206	193	33.8-591

*Excluding outliers values <1 and ≥60.

G. Soil C/N ratios excluding outliers

During the soil health assessment study (Chapter 4), certain soil samples resulted in outlier C/N values, that are typically unlikely to be found in soils (see: Section 4.1). Thus, Appendix Figure 76 displays C/N ratios for farms excluding all outlier values (<1 and ≥60). This is shown for groupings based on urbanisation (A) and management type (B), as well as for upper (1) and lower (2) sampling depths.

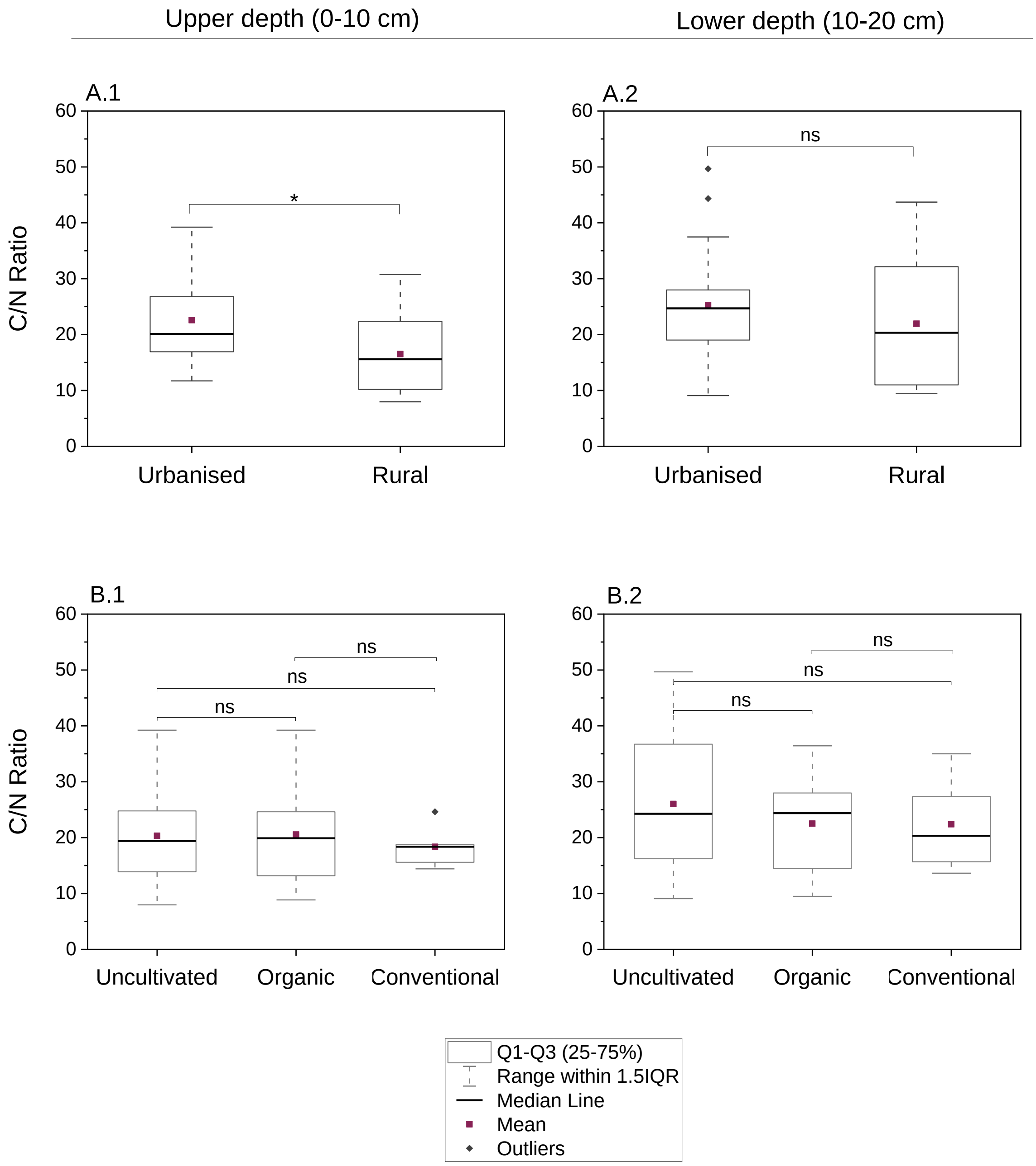

Figure 76 *– Box plots depicting C/N ratios from sampled farm soils, excluding outlier values (<1 and ≥60). Data is grouped based on urbanisation (A), considering urbanised vs. rural soils, or management type (B), considering uncultivated, organically-managed, and conventional soils. Subplots numbered with a (1) refer to soils sampled at the upper depth (0-10 cm) and (2) refer to those sampled at the lower depth (10-20cm). Significant differences between groups are indicated by asterisks; no significant differences are denoted by 'ns.'*

H. Soil health metrics and heavy metal concentrations per farm and landscape type

Appendix Table 82 displays mean values for the main soil health metrics evaluated on each farm as presented in Chapter 4, and Table 83 displays the same information for trace / heavy metal concentrations. Values are respectively presented for the main cultivated spaces (field and polytunnel areas only) and the undisturbed spaces (hedgerows and woodlands) on each farm, separated by a semi-colon. These tables aim to provide comparisons of managed and undisturbed soils on each farm to indicate any potential soil degradation. Percent differences are presented below each value pair. A positive value indicates a higher value seen for the cultivated soil, whereas a negative value indicates a lower value on the cultivated soil, in comparison to the undisturbed soil. Soil parameters tested on samples from upper depths (0-10cm) are indicated by a '(u)', while those tested on samples from lower depths (10-20cm) are indicated by an '(l)'.

Table 82 – Soil health metrics for individual farms, presented as mean values of cultivated and undisturbed soils (respectively) with percent differences

Soil parameter	Unit	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-5 (organic)	R-C-3	R-C-4	R-C-5
BD (u)	g cm^{-3}	0.96 ; 1.02 (-6.4%) [a]	0.84 ; 0.77 (+9.6%)	0.90 ; 0.79 (+14%)	0.77 ; 0.79 (-2.5%)	0.99 ; 0.82 (+21%)	0.69 ; 0.79 (-14%)	0.94 ; 0.76 (+24%)	1.32 ; 0.84 (+58%)	1.41 ; 0.98 (+44%)	1.06 ; 0.78 (+36%)	1.24 ; 0.84 (+48%)
BD (l)	g cm^{-3}	1.24 ; 1.16 (+7.0%)	0.87 ; 0.98 (-11%)	0.92 ; 0.98 (-6%)	0.88 ; 1.0 (-12%)	1.03 ; 0.81 (+26%)	0.85 ; 0.86 (-0.8%)	1.04 ; 0.97 (+7.0%)	1.45 ; 0.86 (+68%)	1.38 ; 1.12 (+23%)	1.11 ; 1.01 (+9.8%)	0.92 ; 0.86 (+6.6%)
WHC (u)	%	68 ; 68 (-0.8%)	74 ; 50 (+49%)	76 ; 74 (+2.0%)	118 ; 68 (+74%)	71 ; 59 (+21%)	79 ; 76 (+3.9%)	61 ; 66 (-8.0%)	54 ; 68 (-20.2%)	52 ; 71 (-28%)	72 ; 70 (+1.9%)	57 ; 68 (-16%)
WHC (l)	%	52 ; 58 (-10%)	72 ; 57 (+26%)	67 ; 66 (+1.4%)	88 ; 63 (+40%)	60 ; 56 (+8.2%)	72 ; 74 (-2.5%)	55 ; 60 (-8.4%)	45 ; 57 (-21%)	49 ; 75 (-35%)	62 ; 66 (-6.6%)	48 ; 57 (-15%)
SOC (u)	mg g^{-1}	58 ; 35 (+65%)	44; 66 (-33%)	39 ; 40 (-2.5%)	92 ; 61 (+50%)	42 ; 45 (-5%)	45 ; 39 (-12%)	35 ; 90 (-62%)	6.3 ; 18 (-65%)	7.8 ; 44 (-82%)	33 ; 76 (-57%)	5.7 ; 18 (-68%)
SOC (l)	mg g^{-1}	37 ; 30 (+22%)	43 ; 32 (+33%)	34 ; 17 (+97%)	77 ; 51 (+51%)	42 ; 47 (-11%)	46 ; 35 (+30%)	36 ; 40 (-10%)	2.0 ; 18 (-99%)	37 ; 36 (+2.1%)	26 ; 87 (-70%)	4.7 ; 18 (-74%)
SOC stock (to 10 cm)	kg m^{-2}	10 ; 7.1 (+40%)	7.0 ; 9.3 (-25%)	6.7 ; 5.6 (+20%)	13 ; 9.1 (+46%)	8.2 ; 5.7 (+46%)	6.0 ; 6.0 (-0.3%)	6.1 ; 14 (-57%)	1.6 ; 2.5 (-38%)	2.2 ; 7.7 (-71%)	6.9 ; 12 (-41%)	1.1 ; 2.5 (-56%)
SOC stock (to 20cm)	kg m^{-2}	19 ; 14 (+36%)	14 ; 15 (-9.0%)	13 ; 7.3 (+73%)	26 ; 18 (+46%)	17 ; 8.3 (+102%)	14 ; 9.2 (+48%)	13 ; 21 (-39%)	3.1 ; 5.3 (-42%)	12 ; 16 (-23%)	12 ; 29 (-57%)	1.6 ; 5.3 (-69%)
N (u)	g kg^{-1}	3.5 ; 1.8 (+102%)	3.4 ; 3.4 (+0.3%)	2.7 ; 2.0 (+34%)	4.1 ; 2.4 (+74%)	2.4 ; 2.0 (+23%)	4.9 ; 4.2 (+19%)	0.95 ; 2.0 (-53%)	0.22 ; 2.4 (-91%)	7.9 ; 11 (-30%)	1.8 ; 3.0 (-39%)	0.28 ; 2.4 (-88%)
N (l)	g kg^{-1}	1.6 ; 1.7 (-5.2%)	3.0 ; 1.4 (+114%)	2.1 ; 8.3 (-75%)	3.0 ; 1.9 (+59%)	1.9 ; 1.84 (-89%)	5.2 ; 3.1 (+68%)	1.1 ; 0.95 (+17%)	0.17 ; 2.0 (-92%)	0.47 ; 14 (-97%)	1.6 ; 1.7 (-4.3%)	0.11 ; 2.0 (-95%)
C/N (u)	Ratio	24 ; 21 (+13%)	13 ; 19 (-32%)	16 ; 24 (-33%)	22 ; 29 (-25%)	18 ; 23 (-20%)	10 ; 10 (+1.4%)	147 ; 66 (+124%)	25 ; 11 (+133%)	74 ; 17 (+341%)	19 ; 158 (-88%)	25 ; 11 (+133%)
C/N (l)	Ratio	24 ; 20 (+20%)	15 ; 24 (-38%)	19 ; 29 (-36%)	28 ; 32 (-11%)	20 ; 22 (-9.8%)	9 ; 12 (-20%)	106 ; 54 (+97%)	35 ; 10 (+260%)	82 ; 26 (+218%)	16 ; 107 (-85%)	35 ; 10 (+260%)
P (u)	mg kg^{-1}	1,436 ; 560 (+156%)	911 ; 453 (+101%)	1,714 ; 937 (+83%)	1,274 ; 489 (+160%)	946 ; 383 (+147%)	1,628 ; 1,060 (+54%)	1,371 ; 459 (+199%)	477 ; 1,227 (-61%)	430 ; 408 (+5%)	503 ; 482 (+4%)	591 ; 1,227 (-52%)
P (l)	mg kg^{-1}	1,106 ; 592 (+87%)	927 ; 371 (+150%)	1,752 ; 751 (+133%)	880 ; 379 (+133%)	829 ; 337 (+146%)	1,312 ; 875 (+50%)	1,095 ; 377 (+191%)	475 ; 943 (-50%)	402 ; 326 (+23%)	475 ; 417 (+14%)	527 ; 943 (-44%)

[a] Values are displayed as averages: cultivated soils ; undisturbed soils. Percent differences are provided in parentheses as increases (+) or decreases (-) from undisturbed to cultivated values.

Table 83 – *Trace / heavy metal concentrations for individual farms, presented as mean values of cultivated and undisturbed soils (respectively) with percent differences*

Metal	Unit	U-O-2	PU-O-1	PU-O-2	PU-O-3	PU-O-4	R-O-1	R-O-2	R-C-5 (organic)	R-C-3	R-C-4	R-C-5
As (u)	mg kg^{-1}	10.8 ; 9.73 (+11%) [a]	14.3 ; 19.4 (-26%)	18.9 ; 36.3 (-48%)	9.16 ; 10.4 (-12%)	7.59 ; 7.00 (+8.4%)	8.79 ; 8.82 (-0.3%)	4.02 ; 2.08 (+93%)	8.88 ; 10.9 (-18%)	6.75 ; 6.56 (+2.9%)	146 ; 79.4 (+84%)	10.7 ; 10.9 (-1.3%)
As (l)	mg kg^{-1}	11.1; 10.8 (+3.3%)	14.6 ; 11.6 (+27%)	20.4 ; 22.3 (-8.7%)	10.6 ; 9.10 (+16%)	7.53 ; 7.23 (+4.2%)	7.96 ; 8.05 (-1.2%)	3.43 ; 2.39 (+44%)	9.08 ; 9.01 (+0.7%)	6.15 ; 6.96 (-12%)	132 ; 89.8 (+48%)	11.0 ; 9.01 (+22%)
Cd (u)	mg kg^{-1}	0.86 ; 0.57 (+51%)	0.37 ; 0.28 (+32%)	0.44 ; 2.95 (-85%)	0.34 ; 0.13 (+167%)	0.33 ; 0.29 (+17%)	0.45 ; 0.40 (+13%)	0.18 ; 0.07 (+178%)	0.20 ; 0.21 (-1.5%)	0.18 ; 0.15 (+18%)	0.47 ; 0.28 (+71%)	0.15 ; 0.21 (-29%)
Cd (l)	mg kg^{-1}	0.76 ; 0.58 (+30%)	0.40 ; 0.23 (+72%)	0.47 ; 0.40 (+15%)	0.38 ; 0.22 (+75%)	0.38 ; 0.17 (+119%)	0.29 ; 0.26 (+10%)	0.14 ; 0.06 (+131%)	0.17 ; 0.12 (+43%)	0.16 ; 0.12 (+37%)	0.41 ; 0.36 (+14%)	0.13 ; 0.12 (+11%)
Cu (u)	mg kg^{-1}	33 ; 19 (+76%)	31 ; 39 (-19%)	26 ; 44 (-41%)	38 ; 21 (+78%)	28 ; 18 (+55%)	25 ; 19 (+33%)	30 ; 9 (+226%)	18 ; 21 (-16%)	26 ; 21 (+27%)	100 ; 52 (+91%)	14 ; 21 (-35%)
Cu (l)	mg kg^{-1}	26 ; 19 (+37%)	31 ; 18 (+69%)	25 ; 23 (+12%)	33 ; 20 (+66%)	31 ; 19 (+60%)	22 ; 19 (+18%)	28 ; 7 (+298%)	15 ; 15 (-16%)	23 ; 21 (+27%)	95 ; 60 (+91%)	14 ; 15 (-35%)
Fe (u)	g kg^{-1}	18.7 ; 16.3 (+15%)	21.1 ; 20.9 (+0.9%)	19.0 ; 26.6 (-29%)	16.7 ; 19.5 (-15%)	19.1 ; 18.3 (+4.5%)	18.5 ; 22.4 (-17%)	12.7 ; 8.35 (+52%)	12.8 ; 13.9 (-7.9%)	9.38 ; 8.21 (+14%)	34.4 ; 34.4 (0.1%)	14.7 ; 13.9 (+6.2%)
Fe (l)	g kg^{-1}	20.8 ; 17.5 (+19%)	18.2 ; 24.1 (-25%)	21.2 ; 19.7 (+7.4%)	18.5 ; 19.0 (-2.9%)	19.8 ; 22.0 (-10%)	17.8 ; 20.5 (-13%)	11.1 ; 9.13 (+22%)	13.7 ; 11.0 (+25%)	8.08 ; 7.68 (+5.2%)	35.7 ; 36.5 (-2.2%)	15.8 ; 11.0 (+44%)
Pb (u)	mg kg^{-1}	133 ; 171 (-22%)	89 ; 108 (-17%)	272 ; 443 (-39%)	72 ; 66 (+8.7)	75 ; 45 (+68%)	55 ; 38 (+45%)	23 ; 34 (-32%)	17 ; 17 (+3.1%)	100 ; 61 (+65%)	107 ; 62 (+71%)	15 ; 17 (-12%)
Pb (l)	mg kg^{-1}	199 ; 110 (+80%)	81 ; 92 (-13%)	267 ; 333 (-20%)	76 ; 79 (-4.1%)	85 ; 46 (+84%)	55 ; 40 (+39%)	19 ; 32 (-41%)	17 ; 13 (+27%)	71 ; 60 (+17%)	115 ; 64 (+79%)	15 ; 13 (+9.1%)
Zn (u)	mg kg^{-1}	260 ; 251 (+3.8%)	112 ; 70 (+60%)	122 ; 187 (+-35%)	124 ; 106 (+17%)	123 ; 100 (+24%)	103 ; 83 (+25%)	72 ; 31 (+131%)	36 ; 49 (-28%)	48 ; 42 (+15%)	295 ; 177 (+66%)	83 ; 49 (+68%)
Zn (l)	mg kg^{-1}	139 ; 119 (+17%)	109 ; 68 (+61%)	124 ; 108 (+15%)	115 ; 105 (+10%)	154 ; 67 (+131%)	64 ; 116 (-45%)	60 ; 38 (+57%)	33 ; 36 (-8%)	41 ; 36 (+11%)	277 ; 194 (+43%)	45 ; 36 (+24%)

[a] Values are displayed as averages: cultivated soils ; undisturbed soils. Percent differences are provided in parentheses as increases (+) or decreases (-) from undisturbed to cultivated values.

I. Qualitative interview guide

This appendix provides detail to the guiding questions used throughout the interviews with individuals receiving food support from Foodhall during the COVID-19 pandemic, with results of this project presented in Chapter 5. The following questions were used to inform conversations with participants, but interviews were semi-structured and thus this does not provide an exhaustive list of each question asked.

Interview Guide:

Information about Foodhall

- How did you learn about Foodhall?
    - *Probe*: When did you learn about Foodhall (before or after COVID-19 lockdown)
    - *Probe*: Did you interact or visit the Foodhall before COVID lockdown?
- What do you know about Foodhall?

Community through Contacts

- Have you made any friends through Foodhall?
    - *Probe*: Do you see them outside Foodhall? Do you speak often?
    - *Probe*: Did you meet them now during lockdown?

Community through Sense of Belonging (participation & contribution)

- Do you ever volunteer at Foodhall?
    - *Probe:* Have you ever thought about it? Why or why not? Have you ever volunteered before?
- Do you feel like you could contribute to the Foodhall project?
    - *Probe:* What roles interest you at Foodhall? What barriers do you see in participating?

Shielding Screening

- During the COVID-19 lockdown, have you been shielding (i.e., did you receive a letter from the government advising you to shield), self-isolating, or not self-isolating?

Sources of Food

- Tell me a little bit about where you mainly get your food from.
    - *Probe*: For example, can you tell me where your food came from each day last week.
    - *Probe:* Was this the same before COVID or has this changed? Why has it changed?
- Where would you say most of your food comes from?
- Have you had to access charity-based food services (i.e., food banks) before COVID?
    - *Probe:* how frequently, i.e., how many times per week or month?
    - *Probe*: If not, when did you start accessing charity-based food services?

Experiences in Accessing Food

- Walk me through your experience of the different ways you obtain food or the different services you use. What is the process for obtaining food? How do you feel about using each service?
    - *Possible probes:* Do you have any preferences in terms of the food services available to you? What is important to you when choosing between different services?

Food Insecurity Screening (USDA and DWP)

"I'm going to read you four statements that people have made about their food situation. For each statement, please tell me whether the statement was often true, sometimes true, or never true for your household in the last month before the UK COVID-19 lockdown (before last 2 weeks of March)."

- 1.) "We worried whether our food would run out before we got money to buy more." Was that often, sometimes, or never true for you in the past month before the COVID lockdown in March?
- 2.) "The food that we bought just didn't last and we didn't have money to get more." Was that often, sometimes, or never true for you in the past month before the COVID lockdown in March?
- 3.) "We couldn't afford to eat balanced meals." Was that often, sometimes, or never true for you in the past month before the COVID lockdown in March?
- 4.) In the past month before the COVID lockdown in March, did you or other adults in the household ever cut the size of your meals or skip meals because there wasn't enough money for food? (Yes/No)
- Did any of these answers change during the COVID-19 lockdown? Which ones?
- *Probe* (depending on answers): What do you do if you feel that you are running out of money to purchase food?

Loneliness Screening (3-Item UCLA Loneliness Scale)

"Please answer how you felt in response to the following questions before the COVID-19 lockdown?"

- 1.) How often do you feel that you lack companionship? Hardly ever or never, Some of the time, or Often?
- 2.) How often do you feel left out? Hardly ever or never, Some of the time, or Often?
- 3.) How often do you feel isolated from others? Hardly ever or never, Some of the time, or Often?
- Did anything change at all during the COVID-19 lockdown? How so?

Social Network Screening (Community Life Survey)

- If you need help, are people who would be there for you?
- If you want company or to socialise, are there are people you can call?
- Do you have someone you can really count on to listen when you need to talk?
- How often do you communicate with your family / friends per week?
- How do you communicate with them?
    - *Probe* for all questions if they say no: why?

General Screening

- Age, gender, ethnicity, number of people in household / household situation, employment status, and income source(s).

Interview Wrap-Up

- What do you see as the most positive experience you've had during COVID?
- Is there anything else you would like to share?
- Are you still okay with me using the information you have provided, as we outlined in the consent form at the beginning of this call?