Agricultural Systems 234 (2026) 104668

Contents lists available at ScienceDirect

Agricultural Systems

journal homepage: www.elsevier.com/locate/agsy

Research Paper

A blueprint of initial LCA in Agri-food production systems: Practical recommendations for crop and livestock production systems[☆]

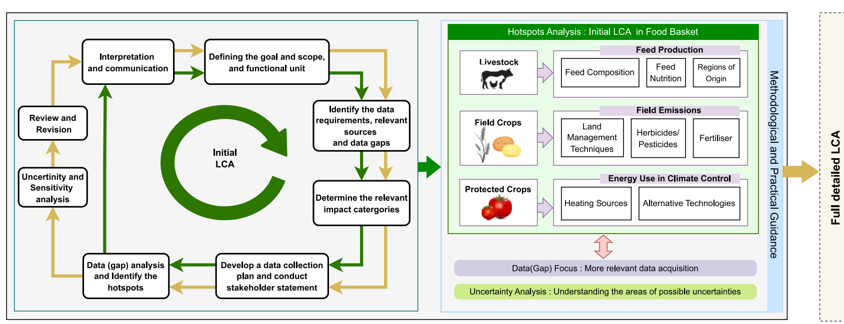
Indika Thushari [a], Asma Jebari [b], Alison Carswell [b], Thomas C. Ponsioen [c], Stephen Morse [d], Richard Murphy [d], Conor Walsh [a,*]

[a] *Natural Resources Institute, University of Greenwich, United Kingdom*
[b] *Net Zero and Resilient Farming Rothamsted Research, North Wyke, United Kingdom*
[c] *FootPrinting (Independent Environmental Consultant), the Netherlands*
[d] *University of Surrey, United Kingdom*

HIGHLIGHTS

- Apart from not conforming to standards scoping LCAs share similar challenges to full studies.
- Initial agri LCAs can assist in decision making but may entail limitations in data etc.
- A farm level scoping LCA is presented to help illustrate practical guidance for non-LCA specialists.
- An international comparison of post-harvest results for tomatoes shows important trade-offs.
- Methodological recommendations are presented for improving initial agri-based LCAs.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Paul Crosson
Guest Editor: P Goglio

Keywords:
Hotspots analysis
Environmental footprints
Agri-food supply chain
animal and plant-based foods
Groceries-food-basket LCA
Scoping LCA

ABSTRACT

CONTEXT: Scoping or initial life cycle assessments (LCAs) occur in the absence of conformity to existing standards and often contend with limited data. This is particularly the case for agricultural food value chains. Nonetheless, such studies can play a valuable role in delivering policy-relevant insights and serving as a precursor for a more complete analysis. However, in practice, such studies may be conducted within the framing of larger projects and be undertaken by subject specialists rather than LCA practitioners.
OBJECTIVE: This study seeks to bridge a gap in the literature to provide practical guidance on methodological pitfalls that may complicate the development of initial LCAs to the point of a minimal viable product (i.e., usable insights or a functioning platform for a more targeted analysis). In this case, our ideal target audience is subject specialists in the agri-food sector, but not necessarily LCA practitioners.
METHODS: This study undertook a structured review of some of the requirements of LCA through the lens of the stages of an LCA and framed by considerations of what a minimum viable product needs. This was supported by the generation of illustrative (plant and animal-based) LCA results produced as part of a commodity 'food

Abbreviations: AFPS, Agri-Food Production System; ABFP, Animal-Based Food Product; FU, Functional Unit; GWP, Global Warming Potential; ISO, International Organization for Standardization; LCI, Life Cycle Inventory; LCA, Life Cycle Assessment; LUC, Land Use Changes; PBFP, Plant-Based Food Product.

☆ This article is part of a Special issue entitled: 'Agricultural LCA methods' published in Agricultural Systems.

* Corresponding author.
E-mail address: c.walsh@greenwich.ac.uk (C. Walsh).

https://doi.org/10.1016/j.agsy.2026.104668
Received 24 March 2025; Received in revised form 9 December 2025; Accepted 3 February 2026
Available online 7 February 2026

basket'. While these terminate at the farm gate based on UK conditions, another illustrative comparison (UK and Spanish tomatoes) is compared up to a substitutable point along the value chain, i.e., a UK distribution centre.
RESULTS AND CONCLUSIONS: Within this work, proposed minimum data requirements are presented along with results for several case studies demonstrating important environmental hotspots. The results demonstrate the importance of interpreting impact hotspots, showing how most commodities will demonstrate life cycle stages that are significant for most environmental impact categories, but not all. For example, in the case of glasshouse-produced tomatoes and broiler chickens, the production of heat and feed is seen as crucial for many impact categories such as Global Warming Potential (GWP), whereas field/farm-based leachate and litter management dominate eutrophication and acidification impacts. Taking a reflective approach, the potential sources of error in developing the exemplar case studies are discussed along with strategies for addressing these.
SIGNIFICANCE: There remains a clear need to formulate practical guidance and recommendations for conducting an initial LCA, highlighting crucial data needs and potential strategies to overcome identified barriers. This paper has shared perspectives on conducting a transparent and reliable scoping study, identifying potential data gaps, limitations, and the complex nature of agricultural value chains, using representative case studies which can be used to further develop more insightful LCAs.

1. Introduction

1.1. Background

Continuous improvement of agri-food production systems (AFPSs) around the globe is essential to mitigate and adapt to climate change, reduce biodiversity loss, and achieve both sustainable and human development goals (Halpern et al., 2019; Notarnicola et al., 2012). In this context, Life Cycle Assessment (LCA) is recognised as a valuable technique for assessing the intensity of environmental burdens and resource use in AFPSs, enabling analysis of trade-offs throughout the product life cycle and beyond. LCA is a complex and iterative technique that provides comprehensive and systematic assessments of environmental impacts, supports decision-making, and facilitates continuous improvement of AFPSs toward more environmentally responsible practices (Notarnicola et al., 2017; Notarnicola et al., 2012; Soussana, 2014).

However, given the complexity of AFPSs, conducting an initial LCA—also referred to as a scoping, streamlining, screening, preparatory, or preliminary LCA (Beemsterboer et al., 2020; Gradin and Björklund, 2021; Hung et al., 2020)—has been recognised as a valuable approach to help with:

a) understanding the key issues of boundary setting,
b) deciding on data collection and analysis, and
c) defining impact categories, thus enabling a more focused and efficient analysis (Barthel et al., 2017; Barthel et al., 2014).

A scoping study refers to an initial stage in the LCA process, acting as a self-contained precursor to a "full" LCA. A full LCA complies with the general International Organisation for Standardisation (ISO) standards (ISO, 2006a) for conducting LCA (ISO, 2006a, 2006b). ISO compliance is important for maintaining rigour but can be time- and data-intensive. An initial LCA is, therefore, part of an iterative process that can identify key stages in the agri-food supply chain contributing to environmental impacts, thereby informing initial strategies for impact reduction and prioritisation around the most significant impacts or benefits associated with the system (Barthel et al., 2014).

Initial LCAs can make valuable contributions to both scientific literature and policy advice when they meet minimum quality standards. However, what these standards entail in practice is not necessarily clearly defined or easily operationalised. The quality of non-ISO-compliant AFPS LCA publications varies considerably, potentially leading to misconceptions.

On the one hand, LCA practitioners face a wide array of challenges in undertaking assessments for products and processes in AFPSs due to regional data unavailability, difficulty in determining boundaries and functional units (FUs), and complex relationships between inputs, environmental conditions, and outputs within multifunctional systems (Notarnicola et al., 2017; Ponsioen and van der Werf, 2017). While these challenges are universal, they are particularly impactful for AFPSs. For instance, the dynamic nature of a herd in an animal production system—where animals contribute multiple co-products to different markets—adds significant complexity to system modelling. Similarly, the spatial and temporal variability in yields, input needs, and environmental impacts inherent to crop production systems further complicates data collection, system representation, and interpretation.

On the other hand, several methodological guidance documents exist to support hotspot analysis in various sectors (Barthel et al., 2017; Barthel et al., 2015; Barthel et al., 2014; Hung et al., 2020). However, there is no globally accepted standard for conducting a scoping analysis on AFPSs that focuses on techniques for working with incomplete or scarce data and creating an inventory that meets minimum data quality requirements (Gradin and Björklund, 2021). Indeed, many initial LCA studies extend beyond what would typically be anticipated for a scoping study. Counterintuitively, the presence of numerous existing studies and data sources can complicate the selection of relevant data or exemplar studies to serve as templates for the scoping study. These sources are often incoherent, opaque, unrepresentative, not entirely applicable to the study, or outdated.

In some cases, the impact assessment results derived from such secondary data sources—Agri-footprint (Blonk et al., 2022; van Paassen et al., 2019), AGRIBALYSE (Colomb et al., 2015; P. Koch, 2015), World Food Life Cycle Database (WFLDB) (Bengoa et al., 2015; WFLDB, 2022), Ecoinvent (B.P. Weidema, 2013; Spindler and Ciroth, 2022)—may be dominated by erroneous inventory parameters (e.g. lack of temporal or geographic representativeness, non-verified assumptions regarding fertiliser and pesticide use, and land use), deviating allocation methods, and generic emission modelling. In some cases, product names in secondary databases match the description but link to a different product used as a proxy, as seen in the AGRIBALYSE database (Colomb et al., 2015).

Therefore, LCA practitioners must be cautious when using secondary sources for processes that are central to the scoping study. Hotspot analysis is crucial, requiring practitioners to recognise erroneous hotspots or those heavily influenced by poor data quality. By doing so, errors can be eliminated, and the quality of important parameters improved.

There is also no widely agreed method for operationalising the findings of hotspot analysis into practicable knowledge and understanding for use by businesses, policymakers, and other AFPS stakeholders. Zampori and Pant (2019) have proposed a method to conduct hotspot analysis—defined as flows that collectively contribute to ≥80% of an impact—consistently. However, in practice, this interpretation is challenging to apply consistently across similar processes and diverse impact categories.

1.2. Literature review

Despite the standardisation efforts by ISO 14040/14044, the framework inherently leaves LCA practitioners with a range of choices which, if not considered properly, can affect the legitimacy of the results (ISO, 2006a, 2006b; Ponsioen and van der Werf, 2017; Weidema, 2014). Several efforts have been undertaken to harmonise these methodological and practical challenges, more specifically in AFPSs (Gheewala et al., 2020; Goglio et al., 2024; McAuliffe et al., 2023a; McAuliffe et al., 2024; McAuliffe et al., 2020; McAuliffe et al., 2018; Notarnicola et al., 2017; Ponsioen and van der Werf, 2017; Poore and Nemecek, 2018). These studies identified recurring methodological and practical issues, including FU definition, improper system boundary setting, inconsistent allocation methods, data gaps, and a lack of coherent tools and datasets for conducting LCA studies. Whilst many of these challenges are well recognised in full-LCAs, they are equally, if not more, pertinent in initial LCAs where simplification often hides methodological compromises that may have been applied.

A more meaningful FU definition reflecting food's multifaceted functions, notably nutrition, social aspects, or cultural functions, is clearly one of the critical aspects in agri-food LCAs (Costa et al., 2020; Fan et al., 2022; McAuliffe et al., 2020; Nemecek et al., 2011; Notarnicola et al., 2017). Moreover, the rapidly evolving nature of technology and methodology further complicates the ability to make accurate predictions even for initial LCAs, as assumptions based on current values on FU, system boundaries, and allocation issues (including in LCI databases) may quickly become obsolete (Notarnicola et al., 2024; Notarnicola et al., 2017; Notarnicola et al., 2012). As literature (McAuliffe et al., 2024; McAuliffe et al., 2023b; McAuliffe et al., 2020) emphasised, global environmental awareness is driving consumer-centric research, which has led to nutritional-LCA, which explores the intersection of environment and nutrition, rather than traditional LCA, which has been used to identify environmental hotspots in the agri-food supply chain. On the other hand, Poore and Nemecek (2018) emphasise that both production efficiency improvements and dietary changes are vital for reducing food's overall impact, offering broader intervention points on system boundaries and defining the FU. Although, as highlighted by other commentators such as McAuliffe et al. (McAuliffe et al., 2023a; McAuliffe et al., 2017), a protein-based indicator ideally needs to consider protein quality. Similarly, Pérez et al. (2024) highlighted the need to recognise the final objective or the system function of focus (i.e., producing fruits, cultivation and area, economic benefits, or nourishing people) to draw valid conclusions in LCA studies. A clear and detailed explanation of the allocation procedures and decisions made following the ISO guidelines, as well as a rationale for prioritising one allocation over another, is essential for transparency in any LCA study (Dominguez Aldama et al., 2023; Pelletier et al., 2015). However, a majority of LCAs do not offer a clear rationale on allocation related to multifunctional systems (Dominguez Aldama et al., 2023; Ponsioen and van der Werf, 2017). Similarly, Wilfart et al. (2021) emphasise the common conflicts of interest in the allocation in the meat supply chain. Even though economic allocation is preferable in many instances in the literature, they highlighted the "inability to choose a single best allocation rule based on scientific and technical arguments alone" ((Wilfart et al., 2021); page 10) and recommended revising the ISO 14040/14044 hierarchy of allocation rules. Ardente and Cellura (2012), also highlighted that even though ISO 14044 guidelines direct economic allocations as one of the last resorts, they simply illustrate properties of complex systems.

A specific inventory dataset is needed to address the specificities of AFPSs, including regional representation (Notarnicola et al., 2024). Notarnicola et al. (2017) emphasise the need for tailored approaches in LCA for food systems due to their inherent complexity and variability. They highlighted the urgent need to improve the quality of LCI data, including in LCA databases, representing the inherent variability of the agricultural systems. Additionally, Poore and Nemecek (2018) meta-analysis provides a crucial grounding for new LCA practitioners by demonstrating the immense variability in environmental impacts even for the same food product thereby underscoring the limitations of using averages. The paper also highlights the necessity of considering multiple environmental indicators simultaneously to avoid problem shifting and provides a vast, harmonised dataset that can serve as a valuable initial LCI resource. Pernollet et al. (2017) showed that limited-depth methods can also remain robust depending on the goal of the study. They developed three simplified methods for LCA of foods and diets and concluded that, compared to a full LCA with inadequate system boundaries, the simplified LCA methods can yield more accurate results at a lower cost of data collection.

A wide array of studies presented initial LCA for the early system planning and design in many fields: biobased material (Heidari et al., 2019), fabric technologies (Dowson et al., 2012), pharmaceutical manufacturing (Budzinski et al., 2022), solid waste management (Wang et al., 2021), and the Food and feed sector (Beemsterboer et al., 2020). Despite the widespread use of streamlined studies, there is a relative lack of literature that discusses the pros and cons of simplification on its own terms, in practice and methodologically (Beemsterboer et al., 2020). The practical challenges faced in initial LCA in different fields have been widely recognised but inconsistently documented (Barthel et al., 2014; Gradin and Björklund, 2021). However, Kiemel et al. (2022) identified five simplification strategies of LCA in the early stage of industrial and product development applications and correlated these strategies with barriers related to time and effort, complexity, and data intensity. Gradin and Björklund (2021) reported ten simplification categories, identified the need for a standardised taxonomy and reporting template for the simplification process. Additionally, (Arzoumanidis et al., 2017; Arzoumanidis et al., 2014) have discussed simplified LCA approaches used in the food sector and highlighted the importance of having dedicated and simplified software tools. Casson et al. (2023) based on a review of 79 tools, recommend approaches towards transparency and strategies to avoid superficial outcomes in simplified LCA that have been developed for non-expert users in the agri-food sector.

Even though direct identification of initial LCAs in the agri-food sector is limited, various publications report initial LCA results and conclusions with the purpose of providing advice to help guide policy development (Adewale et al., 2016; Ferguson and Ramanathan, 2020; Frankowska et al., 2019; Webb et al., 2013; Williams et al., 2010) Any type of simplification increases the uncertainty of LCA results (Liu et al., 2024), and the problem is further exacerbated by the fact that, in many instances, only a qualitative uncertainty analysis is feasible in these initial LCAs (Igos et al., 2019).

1.3. Objective

This paper aims to present practical and methodological recommendations for planning and conducting an initial LCA for AFPSs, focusing on appropriate techniques for working with readily available—but often incoherent—data as a starting point in study design. Specifically, it seeks to share propositions for planning and executing a qualitatively and methodologically coherent initial LCA, with an emphasis on data needs and risks in targeted life cycle stages, particularly in light of data availability and quality limitations.

These propositions are framed within the context of an initial LCA study, outlining associated risks, challenges, limitations, and strategies to overcome them, while focusing on key aspects that support the transition to full LCAs. By way of example, a selection of commonly produced commodities—beef cattle, pigs, broiler chicken (all expressed in liveweight terms), eggs (from layers), ware potatoes, wheat grains, and fresh tomatoes—are included as "cradle-to-farmgate" and "cradle-to-field gate" case studies for livestock and crop products, respectively. These commodities are of international importance in terms of both market demand and their contribution to sectoral emissions (Defra, 2023; Defra, 2022; USDA, 2023).

Table 1
Idealised distinction of an 'initial' LCA and a 'full' LCA.

	Initial LCA	Full LCA
Basic function	To gain insight into the determinants of the environmental impact and its scale	To conclude on the environmental impacts and enhance understanding about the unit processes
Granularity	Broader and more generalised view (dependent on the setting)	Detailed and specific view
Resources and time availability	Less intensive in resources and time (dependent on the context)	Higher resource intensity As data quality needs to be ensured before additional sensitivity/uncertainty analysis
Setting	Broader scope, inclusion of specific and more generalised data and assumptions	More focused scope, detailed data requirement, more comprehensive analysis/ interpretation
Data requirement	Relies on aggregated or secondary data in the first instance. (lower data precision and quality requirements)	Comparatively higher data quality (primary or secondary data) requirements (relatively higher data precision)
Application	Identify major hotspots/ areas for improvement/data gaps/comparisons	Same as scoping study, but expanded role in informing specific sustainable decision-making and marketing positions.
Strengths	Provides insights on potentially important contributors to environmental impact and directions for better judgments on deciding the goal/scope of a full LCA	Provides a more detailed and comprehensive understanding of the uncertainty in data quality and limitations of derived insight/results
Weaknesses	Difficult for definitive claims/ conclusions to be made without additional uncertainty analysis	More detailed data coverage is needed. Takes more time and budget (even after review, conclusions may still be contentious)
Conforming to Existing Standards	Partially conforming to standards (i.e. compliant in some aspects only, but not all)	Complaint in most, if not all aspects of established standards

LCA studies may be conducted by subject-area specialists (e.g. in horticulture or livestock), but not necessarily by LCA specialists. Therefore, the intended audience for this work includes emerging LCA practitioners—such as students, researchers, and consultants—who are seeking practical guidance. In many cases, LCAs are undertaken as part of a broader project. However, limitations due to data gaps and risks associated with the misuse of available data may be intractable in the short term. A better understanding of these issues and potential remedies will help determine whether such an LCA exercise can reasonably meet the needs of the wider project.

2. Methods

Framed according to ISO 14040 and 14044:2006 guidelines (ISO, 2006a, 2006b), the following sections provide the evidence outlining an initial LCA case study of a representative grocery food basket, highlighting limitations, and recommendations whilst demonstrating how to inform a more in-depth analysis towards a full LCA.

2.1. Understanding the initial and full LCAs

The primary objective of doing an initial LCA is to gain an initial understanding of the environmental impact of products, identify the areas with the highest impact (hotspots), compare the impact to reference studies, and prepare for a detailed and comprehensive LCA (Beemsterboer et al., 2020; Klöpffer and Grahl, 2014; Nicoletti and Notarnicola, 1999). A full LCA differs from an initial scoping study in that it requires more structured and detailed reporting of the goal and scope, inventory data and results. This includes a comprehensive data quality assessment, sensitivity and uncertainty analyses, careful interpretation, and a critical external review. These steps are essential in generating conclusions on the hotspots and absolute scores, and when comparisons are explicitly defined in the goal and scope. What distinguishes an initial LCA is that it is often undertaken with limited data and with a focus on inputs and activities that are likely to disproportionally embody burdens. In theory, the setting of an initial and a full LCA also differs in terms of the depth and extent of the assessment (Table 1). However, it should be noted that there is no strict delineation between the level of detail that constitutes a scoping study, as that will depend on the system under review and the goals of the analysis. The only real distinction then is that initial LCAs do not fully conform to the ISO 14040 and 14044 standards; however, such studies can provide valuable insights when they assure transparency and avoid misinterpretation. We argue that many published studies (particularly when LCA is not the primary focus of a study) will fall into this category. A schematic diagram of the initial LCA implementation is presented in Fig. 1.

2.2. Defining the goal and scope

The first stage of any LCA involves an accurate description of the goal and scope in the form of research questions and identifying the intended application and target audience. A limited or internal study may not necessitate ISO compliance, whereas claims of impact reduction ideally should. Goal and scope definition include defining the geographical, process, and temporal boundaries and the system type (ISO, 2006a). For AFPSs, a study examining the impact of agronomic management may not necessitate inclusion or post-harvest on-farm processing. Distinguishing the practices (e.g., transport, processing, storage, etc.) relevant to the study objective will reduce the burden of data gathering and add clarity to the comparison of alternatives. Each AFPS has unique characteristics that must be considered when defining the goal and scope (Notarnicola et al., 2017; Poore and Nemecek, 2018). As suggested by Bessou et al. (2013) LCA will yield more relevant and actionable insights by ensuring that the goal and scope are tailored to the specific AFPS.

The goal of the illustrative case study presented here is to carry out an initial cradle to farm or field gate LCA on some representative animal and plant-based agricultural food commodities produced in the UK, to identify data gaps and potential environmental and agronomic hotspots while highlighting the potential risk of using limited-quality data. Furthermore, this will give the relevant insight for additional quantitative investigation for a more detailed full LCA.

2.3. Defining an appropriate functional unit

In the ISO 14040:2006 standard, the FU is defined as the "quantified performance of a product system for use as a reference unit" (ISO, 2006a). The FU serves as the basis for quantifying flows and impacts, allowing the comparison based on functional equivalency when chosen correctly. A key challenge in defining FU is selecting an appropriate FU for a multi-functional system (with co-products or processes) and determining the influence of this choice on ultimate results (Fan et al., 2022; McAuliffe et al., 2023b; McAuliffe et al., 2024; McAuliffe et al., 2020; Notarnicola et al., 2017). ISO standards specify that FU should be quantifiable and based on a clearly defined goal and scope (Panesar et al., 2017). Imprecise definitions (e.g. not distinguishing dry versus wet material) frustrate informed comparisons. Fig. 2 presents a simplified guidance on deciding a relevant FU.

Mass and volume-based FU are more predominantly used in comparative agri-food LCA studies, instead of using actual functions like nutrient and energy supply, and the economic value (McAuliffe et al., 2020; Ponsioen and van der Werf, 2017). Mass-based FU can be easily

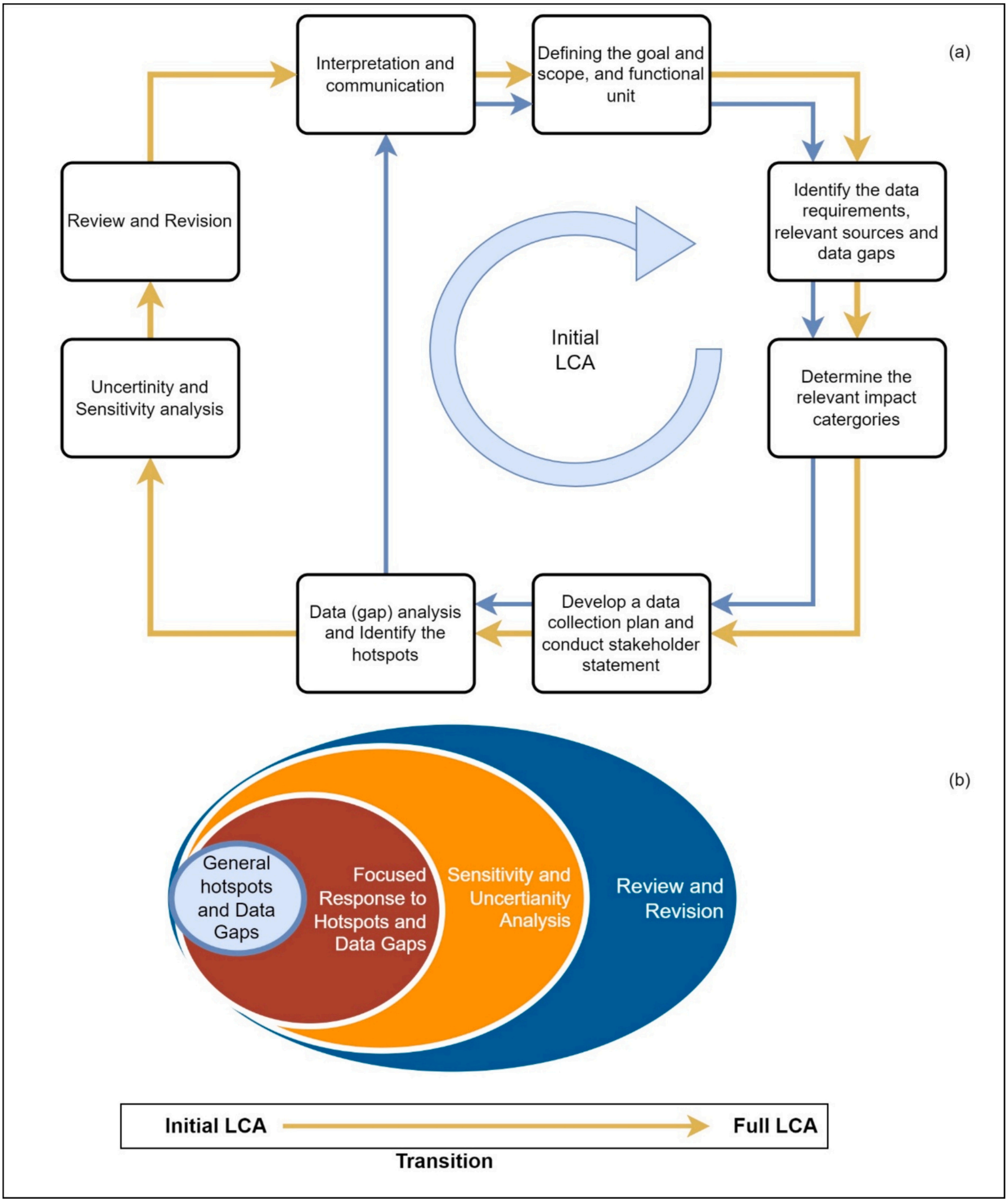


Fig. 1. Conceptual framework for distinguishing key steps of an initial LCA (in blue) and transition from an initial LCA to a full LCA (gold) in terms of LCA stages (a), alternatively represented as a nested process (b).

quantified, and the reference flows in terms of Life Cycle Inventory (LCI) is comparatively straightforward. Consideration of the FU is closely related to consideration of study boundaries and requires awareness of the activities included and not merely the boundary definition (Notarnicola et al., 2017). For example, some commodities, such as orchard fruit, may be cooled on the farm or alternatively transported to a nearby cool house. Therefore, distinguishing between the field and farmgate boundary is important, requiring clarity on the specific value chain endpoint. Also, in some cases, product descriptions may be misleading, for example, 'fresh' may imply recently harvested or produce kept under chilled conditions, whilst 'chilled' may have undergone prior freezing. An inaccurately defined FU can lead to improper selection as an input when extending the study boundary, thereby misrepresenting the actual value chain in a full LCA. Further, several studies discussed the need for multiple FU(s) to capture complex interactions in systems. Studies, such as (Nemecek et al., 2011) demonstrated the use of multiple reference unit to avoid potential misleading in systems comparison in integrated and organic farming systems while (Pérez et al., 2024) explain how the choice of FU can significantly influence the relative carbon footprint of different soft fruit production

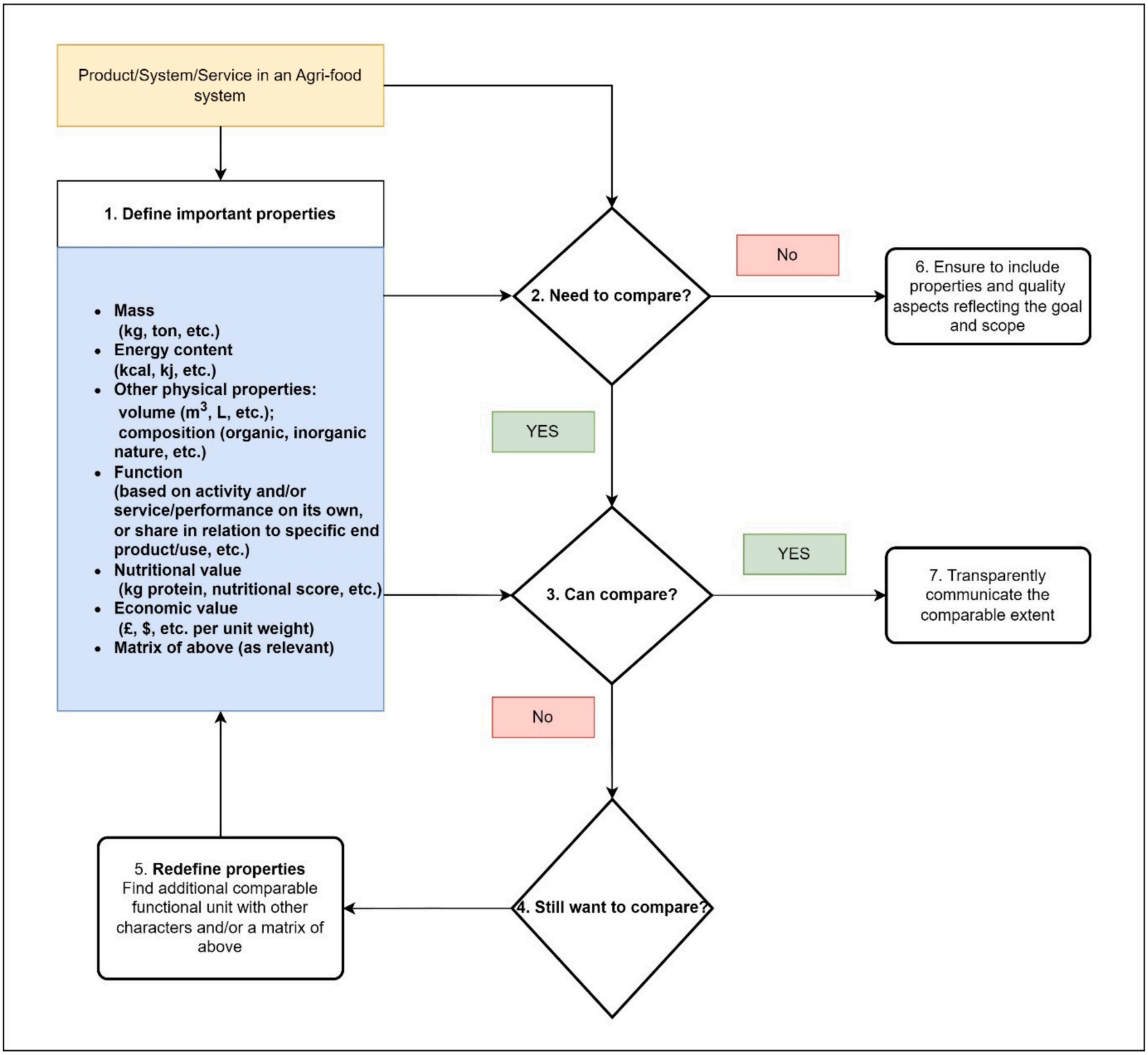


Fig. 2. Simplified guidance on deciding a relevant FU (European Commission, 2010; ISO, 2006b; ISO, 2006a)(European Commission, 2010; ISO, 2006b, ISO, 2006a).

systems, meaning that the preferred production option depends on the metric of comparison. Costa et al. (2020) highlighted the use of multiple FUs to represent multifunctionality in crop rotation, to capture both the outcomes of entire rotation sequences and product footprints, in line with Goglio et al. (2018), who emphasised the importance of distinguishing between product and systems LCA in cropping systems.

In this study, a mass-based FU has been used to compare the environmental burdens of selected commodities due to its simplicity. By way of illustration, the environmental performance of lowland suckler beef calves (with minimal time indoors), indoor pig, and broiler production systems has been evaluated using the FU of 1 kg liveweight at farmgate. Chicken eggs were evaluated using the FU of 1 kg of eggs at the farmgate. Similarly, life cycle environmental burdens and resource use for producing 1 kg of fresh field crops (ware potatoes and wheat) and protected horticultural products (tomatoes) were considered at the field gate, excluding post-harvest processing. These choices reflect the focus of the scoping study on comparing agronomic production systems and merit consideration when interpreting comparative results.

2.4. Determine the co-products, allocation methods in a multifunctional AFPS

According to the hierarchical guidelines in ISO 14044:2006, multifunctionality should be managed by:

(i) subdividing the system to separate processes when feasible,
(ii) using system expansion, which accounts for the impacts avoided by producing co-products that displace the need for other products,
(iii) partitioning based on a physical relationship between inputs and products, if such a relationship exists, and
(iv) other allocation methods, such as economic or energy-based allocation, if no physical relationship can be identified (ISO, 2006b).

While the ISO hierarchy provides a clear framework, several challenges arise when applying these methods to AFPSs (Notarnicola et al.,

Table 2
Allocation methods across food basket case study.

Agricultural commodities/ Multifunctional outputs		Main issue	Allocation method-used in this study	ISO Hierarchy position	Rationale
Agricultural commodity	Main waste				
Beef Cattle. i.e. 2 year old calves sold.	Manure Spent Straw	Herd dynamics at farm gate; System complexity (i.e., beef production involves multiple stages: from breeding till slaughter or distribution) Diet and production system variability	Bio-physical based on live weight at point of sale for beef calves, cows and bulls. Approx. 80% allocated to Beef cattle.	Allocation based on bio-physical conditions	Biophysical allocation helps to trace the flow of biological inputs or functions. It determines how much feed, energy, or emissions go into producing protein or carcass mass.
Live Swine	Manure	The system produces multiple co-products in different stages.	Allocation occurs during piglet rearing based on mass (95% allocation to piglets). Piglets are an input into pig rearing with 100% allocation.	Mass-based bio-physical allocation	The process distinguishes two interconnected stages: - Sow/piglet rearing: in where allocation happens based on biophysical conditions -Gilt rearing in which 100% of the impact is allocated to live pigs (as the mortality rate was sufficiently low to constitute a negligible co-product (similar rate for broilers)).
Chicken Broilers	Manure Feathers	In practical terms, the main co-product is related to mortality, but the assumed mortality rate (0.04) and low economic value resulted in negligible allocation. For higher mortality rates (0.10), this becomes more consequential.			
Eggs (from layers)	Spent hens.	None considered, as a similar reason to the above. Due to the lower economic value of spent hens, all the allocation has been provided to eggs.			
Ware potatoes	Crop waste	None considered as crop waste has negligible economic value.			
Wheat grains	Straw	Fertiliser use Variability of the edaphoclimatic conditions	Economic	Allocation based on the economic value	Wheat grains and straws serve different functions and have different economic values. Given the economic value of grains (92%) and straws (8%) in the market economic allocation considered
Tomatoes-in polytunnel	Crop waste	None considered as crop waste has negligible economic value.			
Tomatoes-in heated greenhouse	Crop waste	None considered as crop waste has negligible economic value.			

2024; Notarnicola et al., 2017; Pelletier et al., 2015; Ponsioen and van der Werf, 2017; Weidema, 2014). However, allocation LCA, given its industrial origins, becomes more complex when multiple (and intermediate) products are involved, as is the case in agricultural systems. Subdividing unit processes in agri-food LCAs is often impractical, as different parts of a plant or animal that result in distinct co-products cannot be easily separated. For example, raw milk, newborn calves, and live weight are all produced by a single animal and cannot be divided into discrete sub-processes.

Biophysical co-product allocation is feasible in many AFPSs when a physical and causal relationship exists between inputs and co-products. However, such approaches have not been sufficiently elucidated, particularly regarding how selected physical parameters accurately represent causal mechanisms in these systems (Mackenzie et al., 2017). For instance, in dairy farming, if allocation is based solely on protein content, the environmental burden of milk may be overestimated, as milk production involves different processes and energy use compared to beef production from the same cow.

In practice, environmental impacts of AFPSs are often apportioned based on the market value per unit of each co-product, multiplied by its yield, and divided by the total revenue. (Ponsioen and van der Werf, 2017) advocate for economic allocation in multifunctional agricultural processes, while (Ardente and Cellura, 2012) identify it as a simplified method for illustrating complex product attributes. However, while economic allocation offers a practical solution, it is sensitive to short-term market fluctuations (e.g. supply and demand, policy interventions), which may not adequately reflect the true environmental cost of production, potentially leading to misestimation of specific co-products.

A purely biophysical approach may allocate a significant portion of the impact to a product with limited utility. Conversely, studies by Chen et al. (2017) and Wilfart et al. (2021) show that economic allocation can ignore resource efficiency and omit resource streams with no economic value. Distinguishing waste from low-value outputs based solely on economic criteria can also lead to inconsistencies with ISO standards, which recommend allocating environmental burdens only among co-products (Wilfart et al., 2021). Therefore, consistent value data—both geographically and temporally—are needed, along with informed judgment to distinguish low-value co-products from waste.

In this study, mass allocation based on livestock liveweight was used to distribute environmental burdens among livestock co-products, as it was the simplest method and aligns with one key input LCI (piglets) that already applied this approach. However, economic allocation was used for one product—wheat—as the co-product (straw) was produced in sufficient quantity to detract from the main function of the production system. Table 2 presents the allocation methods used in this study. Table 1S (supporting materials-A) presents some recommended allocation strategies used for representative multifunctional AFPSs.

2.5. Selection of an appropriate system boundary

System boundary selection involves identifying the potentially relevant unit processes and their flows of energy and matter in relation to the goal of the study. According to ISO 14040, a unit process is defined as the smallest element considered in the LCI analysis for which input and output data are quantified (ISO, 2006a) Within the system boundary, unit processes are represented along with their interrelationships in terms of the flow of inputs, outputs, and intermediate products. In some cases, unit processes may be excluded from the system boundary if they are deemed to have an insignificant impact on the results (ISO, 2006a).

Decisions regarding inclusion or exclusion should be based on identifying the point at which one system transitions into another. A key risk arises from overly constrained boundaries, which may omit important trade-offs. For example, excluding post-harvest activities may

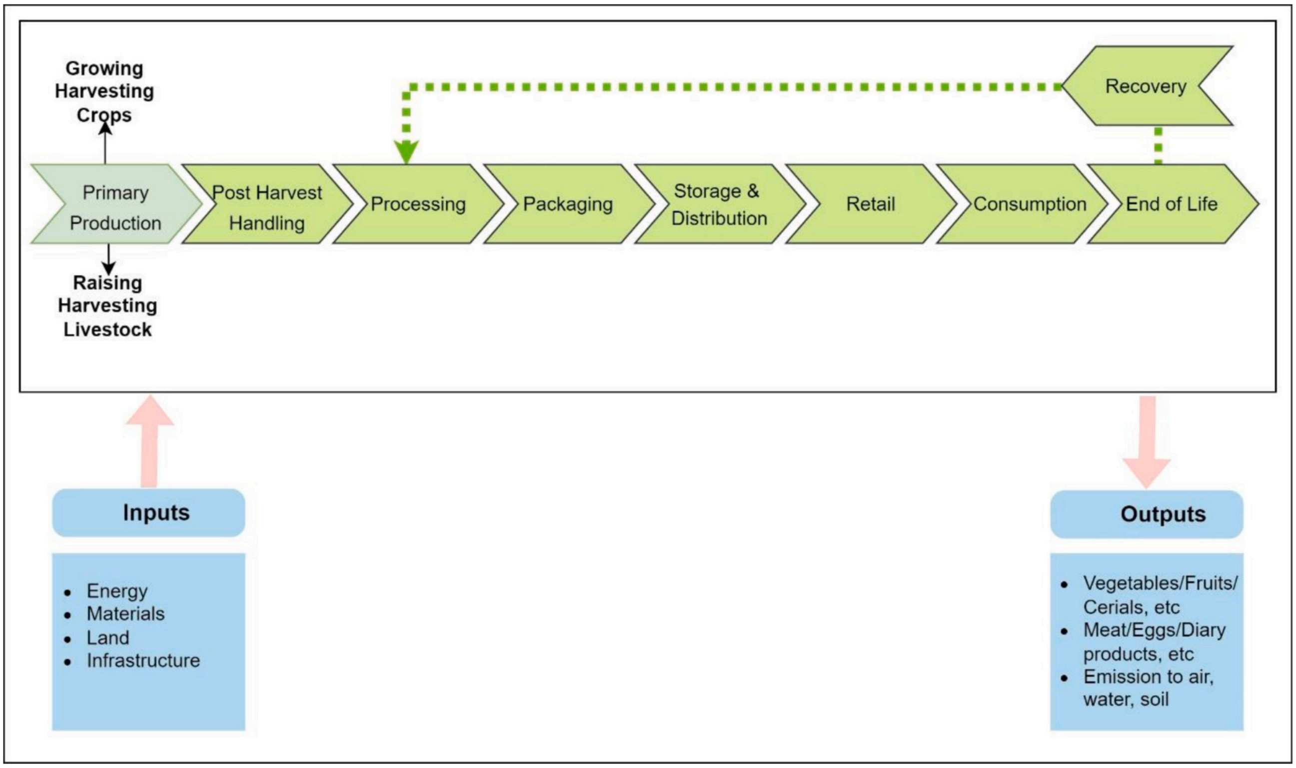


Fig. 3. a: Value chain of a typical AFPS.
b: An indicative system boundary of the considered ABFP and PBFP systems. Upstream and downstream farm activities are different for each crop and livestock production system considered. The boundary in this case terminates at the farm gate for livestock and the field gate for crops. For our case study, additional activities such as chilled storage are excluded but will fall within the boundary of other studies.

simplify the isolation of field-level impacts, but it does not reveal whether these impacts are offset or exacerbated by subsequent activities. Ideally, boundary delineation should distinguish separate value chain stages that can be examined both independently and collectively. In such cases, a study may separate field-level activities and post-harvest processing as distinct but interconnected value chain stages.

Reviewing existing inventories for similar products can help determine which processes and flows should be included in a particular study, which flows appear negligible, or where omissions in individual studies can be compensated through amalgamation. The study boundary ultimately determines the types of questions that can be addressed. For example, questions related to import substitution require consideration of processing, losses, and transport up to the point at which comparable value chains intersect.

In the case studies examined here, a selection of widely consumed conventional UK animal-based food products (ABFPs) and plant-based food products (PBFPs), considering both market demand and contribution to the sectoral emissions, was considered. Beef cattle, Swine, chicken (broilers), eggs from layers, fresh ware potatoes, fresh wheat grains, and fresh tomatoes were included, and the boundary was fixed from 'cradle to farm' and 'cradle to field gate' for the identification of major hotspots for livestock and crop commodities, respectively. The farmgate was chosen as a boundary for the livestock case studies as they are initial, agronomic focused analyses, being precursors to subsequent expansion, which will include the additional stages. This also allows comparison on the basis of agronomic practices. As the livestock case-study reflects to the period where the animal has reached a sufficient age to be processed, it includes time on the farm (and off the field) but excludes additional processing steps.

Furthermore, some practices—such as chilled storage—may occur either on or off the farm, further complicating comparisons. For this reason, the field gate was used as the boundary for crops in which no additional chilling or similar post-harvest processes were included (unlike studies such as (Williams et al., 2006). This approach aligns with the wheat case study, which reflected the temperature conditions during the harvest study period (2021/22). The recorded moisture content for that year (13.5–14.5%) suggested that minimal additional drying was necessary (AHDB, 2022a), although some regions may require additional grain cooling.

Therefore, what is separated by the 'farm' boundary requires careful consideration when comparing nominally equivalent FUs. In the case of tomatoes, two production systems—UK-based heated glasshouses and Spanish unheated polytunnels—were considered to reflect the need to account for differing impact categories across production systems. In support of this, we argue that, where possible, the FUs should include (at least initially) the quantity, quality, and position of the product within the value chain.

A typical value chain of an AFPS will extend beyond the farm gate and is presented in Fig. 3a, whilst the system boundary of the considered case study is shown in Fig. 3b. The main upstream activities considered were the production of agricultural inputs (such as fertiliser, pesticides, herbicides, and feed compounds), energy, machinery, and housing. The intercontinental and international transportation of the inputs has been considered using waterways (shipping), where a feasible inland route is not available. For domestic transportation, trucks and (at the farm/local level) tractors of different capacities have been considered for longer and shorter distances, respectively. The input of energy at various stages of the production chain is in the form of natural gas, diesel, and electricity, as per the requirements and availability for each individual unit process. The energy is used in various forms, such as electricity (for ventilation, lighting, and water pumping), natural gas for heating, and diesel (for agricultural inputs and feed production, transport). For

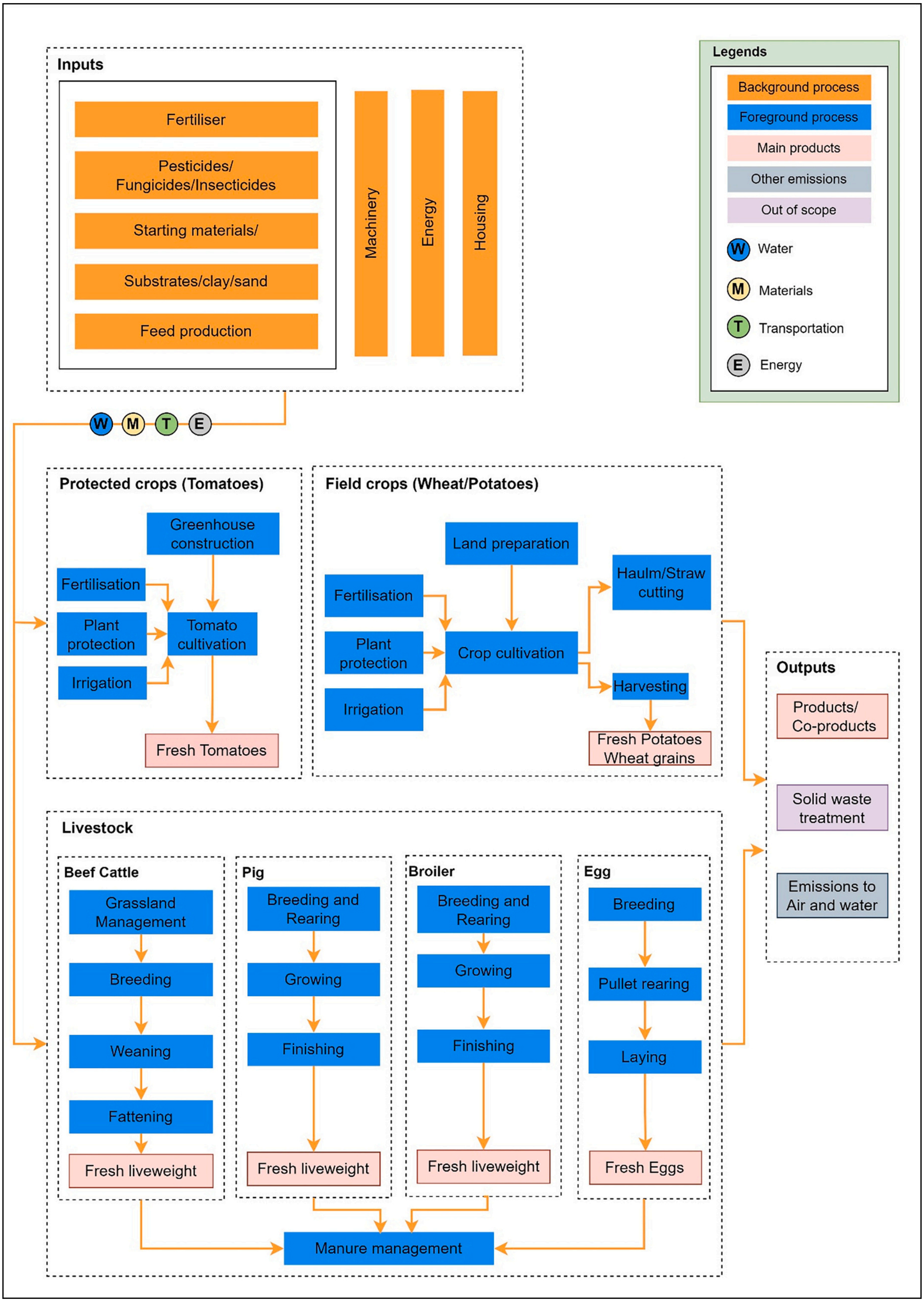


Fig. 3. (*continued*).

example, an additional comparison of Spanish and UK tomatoes, including post-harvest chilled storage and transport to a regional distribution centre in the UK.

This means that insights arising from such a comparison should be viewed with caution when there is likely to be a significant subsequent processing step, as the cradle to gate may reflect a comparatively smaller

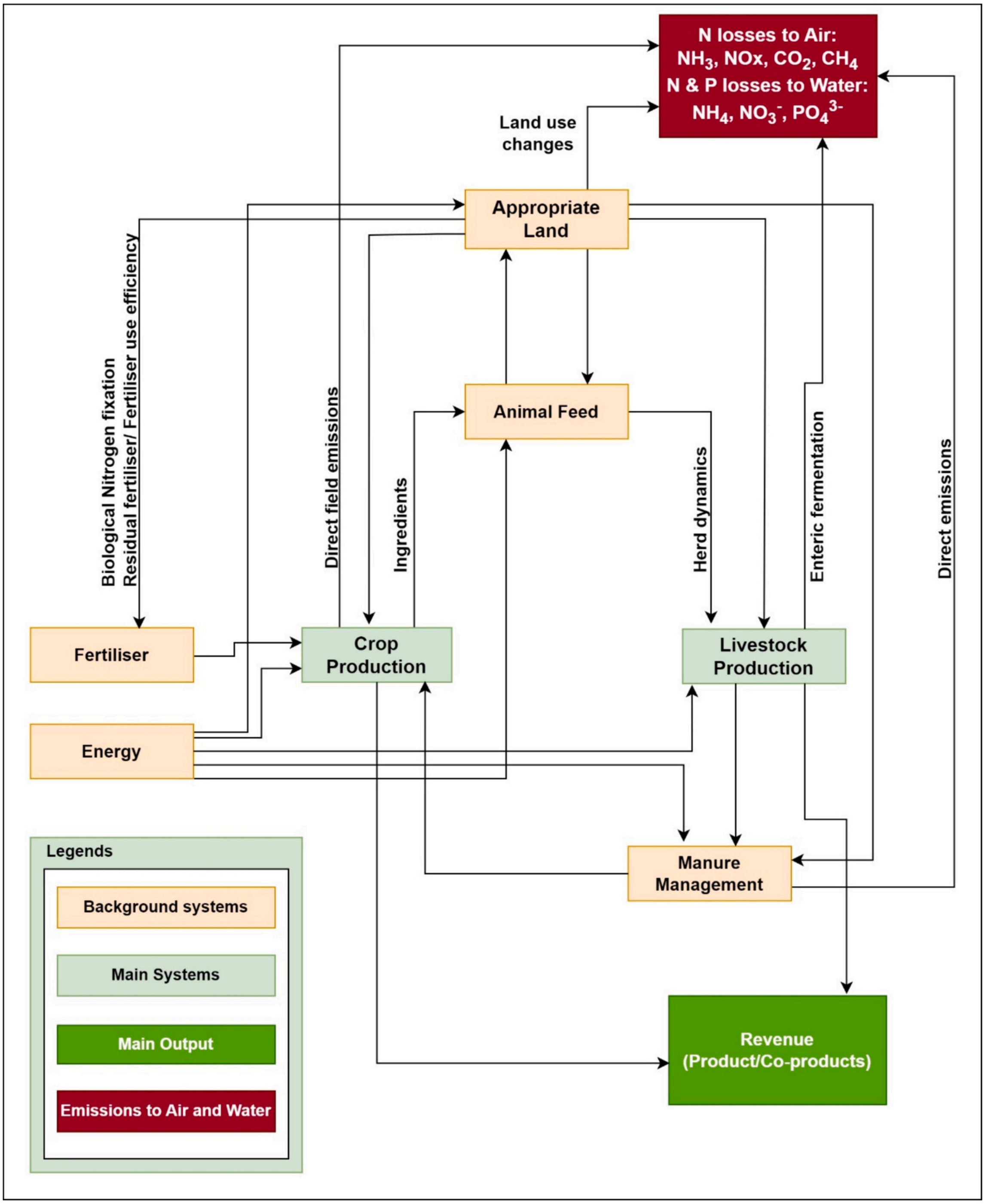


Fig. 4. Direct and indirect relationship between various factors in a typical AFPS up to the primary production (Farm gate).

"hotspot" compared with other processes. Whilst comparison of farm level results is legitimate, it is important not to extrapolate to comparison at the distribution and/or consumer level.

2.6. Determination of data requirements for inventory generation

Determining the data requirements early in the study is vital, as it allows data collection efforts to be focused on the most important variables. These requirements will need to be re-evaluated and possibly revised during the interpretation phase. Existing studies (and exemplar LCIs) with an equivalent FU, boundary, and setting can serve as templates for data acquisition, including guidance on the units in which such information may be expressed. However, this should be approached with caution, as each study will have distinct objectives and boundaries. Developing a data collection plan is critical for obtaining accurate and reliable data (Klöpffer and Grahl, 2014). Depending on the scope, this could include information on the data sources to be used, the methods for collecting data, and the quality assurance methods (Barthel et al., 2014). A clear delineation between crude material inputs (intermediary and final products), co-products, and waste (Amicarelli et al., 2021) by receiving environment and mechanism (e.g., runoff, leachate) will assist in developing an appropriate data gathering template. An important consideration is identifying which outputs have a direct causal/stoichiometric relationship to the inputs (Fig. 4).

Table 3
Commonly found sources of error/uncertainty in the secondary data for studies terminating at the farm gate.

System	Potential sources of error/uncertainty
Crop production	o Incorrect reporting/assumptions of • Fertiliser use over/under-estimation when estimating application solely based on plant/soil needs • Higher level/Tier 1 fertiliser emission modelling based on simple assumptions of N-N_2O emission factor • Failure to consider CO_2 emissions due to lime use • Discounting non-representative use of pesticides, sometimes including prohibited substances • Assuming a higher or lower yield or using an average yield when seeking to reflect specific conditions • Confusing yield in dry and wet weight (and units). • Level of on-farm, post-field losses • Omitting the impact on crop-residue management • Omitting the impact of soil C emissions and land occupation (e.g. does the crop grown on peatland soils have an associated C loss) • Unrealistic amount of diesel use based on aggregated estimate as opposed to individual activities • Outdated/non representative energy use in heated greenhouse horticulture, including grid emission factors • Omitting impact of nutrient losses and emissions through volatilisation, leaching etc.
Livestock production	o Incorrect allocation, such as: • Deviating allocation methods for co-products such as protein allocation in case of sheep farming (e.g. inconsistently applying protein-based allocation to wool and animal live weight) • Handling manure as a co-product when it needs further treatment to render it usable. • Biophysical allocation in case of dairy farming • Outdated/non representative price data for calculating economic allocation factors o Use of non-representative animal feed compositions, including consideration of land use changes (LUCs) o Non-representative or inaccurate representation of manure management techniques o High-level/Tier 1 and/or outdated/deviating manure emission modelling o High-level/Tier 1 modelling of enteric fermentation o Omitting the impact of nutrient losses and emissions through volatilisation, leaching etc.
General	o Misleading product names (product names appearing to represent the description, but then linking to a completely different product that is apparently used as a proxy) o Ignoring mass balance (losses and gains in unit processes) o Lack of full consideration for possible co-products o Lack of full consideration of relevant upstream and downstream processes

Data collection can span surveys, interviews, and field measurements, in addition to primary sources (e.g., market data reports) and secondary data gathered through literature reviews (e.g., national statistics, scientific research) or existing databases (Notarnicola et al., 2024; Notarnicola et al., 2017; Notarnicola et al., 2015). Stakeholder engagement is important, as it can help identify potential data gaps and provide feedback on outcomes (Barthel et al., 2014). Stakeholders in a typical AFPS may include farmers, processors, consumers, and other experts (see Table 2S in the supporting materials-A).

In the LCI phase, environmental loadings in the form of resource use, emissions, and waste are accounted for in each of the unit processes considered within the system boundary. The reliability and interpretation of the results depend on data quality (e.g., accuracy, specificity, representativeness). Errors in data—particularly when using secondary or unverifiable sources—and assumptions can lead to inaccurate results and conclusions. Table 3 presents potentially misleading parameters associated with the use of secondary data for activities focused on the farm or field gate. While these considerations are relevant for all agricultural LCAs, initial LCAs may be particularly susceptible due to limited time and resources.

It should be noted that some of these issues may be intractable or unavoidable in settings with limited data. Therefore, acknowledging these limitations may prompt a redesign of the LCA within the project or prioritise additional data gathering efforts.

The structure of data collection will be informed by the boundaries (and assumptions) of individual product stages; for PBFPs (crops), data are often collected in terms of flows per unit of area, whilst for ABFPs (ruminants), an analysis may be framed at the individual herd level. This entails additional risk when, for example, data on the material inputs into the field are considered separately to outputs like yield. For the case study presented here, a number of existing inventories were used to provide a template for data collection for inputs and outputs. In addition, databases, such as Agri-footprint 6 (Blonk et al., 2022), ecoinvent 3.8 (Wernet et al., 2016) and World Food LCA Database (WFLDB) processes (Nemecek et al., 2019) were used for production inputs and other background data (See Tables 1–8S in the supplementary materials-B for LCIs).

2.6.1. *Inclusion of upstream activities*

The major sources of environmental burdens from upstream activities are manufacturing of fertilisers and other agricultural inputs, production of concentrate feed, land use change (LUC), supply of (fossil and renewable) sources of energy, and generation of electricity. Identifying and categorising all relevant agricultural inputs, such as fertilisers, seeds, pesticides, energy and water (including flows from the Ecosphere) is a precursor to their quantification.

For the presented exemplar case study, all the important upstream activities data have been collected from the relevant literature pertaining to or near the representative UK system. For important inputs such as concentrate feed, the composition for pig, beef, broilers, and layers production have been taken from literature (Casey and Holden, 2025; Hall et al., 2018; Halpern et al., 2019; Kebreab et al., 2016; Leinonen et al., 2014). Soybeans are a major ingredient in livestock compound feed imported into the UK from other countries. Approximately 70% of the soybean meal in the UK is imported mainly from Argentina and some from Paraguay. Whereas whole soybeans and soybean oil are mainly imported from Brazil and the Netherlands, with 60 and 70% of its total requirement, respectively (Efeca, 2019). Therefore, Argentina, Brazil, and the Netherlands have been chosen as importing partners of the UK for soybean meal, whole soybeans, and soybean oil, respectively. Similarly, consideration of regional conditions is necessary to clarify available transport links and distances between discrete stages. This includes whether a specific transport connection, such as rail, is viable in the region of choice. Failure to consider specific input parameters will risk underestimating impacts. For example, in the case of broiler chicken, approximately 48% of the global warming impact score is attributable to LUC.

2.6.2. *Inclusion of on-farm activities*

The nature of the product/system, specifically the activities associated with different production systems (e.g., open field vs indoor) will determine the data requirement and the material/energy flows that need to be quantified at the farm level. For example, PBFPs may need explicit consideration of establishment activities, such as in the case of glasshouse production. The consideration of field or horticultural systems, whilst encompassing different activities (such as the infrastructure) share common data needs in terms of land, fertiliser, energy, and water use. Indoor system energy needs are likely to be determined by heating requirements, whereas energy needs for field systems will depend on the extent of mechanisation, with post-field cool storage being commodity, value-chain, and region specific. In cases with multiple harvests, it will be necessary to compare inputs (resource use) and outputs (yield and emissions) across the same time boundary. (As mentioned for the crop

Table 4
Examples of relevant mid-point impact categories and related activities in AFPS. Each of these impact categories will reflect a different environmental burden, based on Recipe 2016 (Huijbregts et al., 2017) and IPCC, 2021 (IPCC, 2021).

Impact category	Unit	Description/ Function	Related activities in a typical AFPS
Climate change – Total, Fossil, Biogenic and land use	kg CO_2-eq	GWP, an indicator of integrated infrared radiative forcing increase of greenhouse gases (GHG) (Stocker et al., 2014). Reflects contribution to global burden of a key driver for climate change. Not to be confused with the impact of climate change.	• Manure management • Enteric fermentation • Fertiliser production and use • Deforestation and LUC • Energy use • Crop residue burning • Agrochemical production
Eutrophication – Freshwater Marine water and Terrestrial	kg PO_4^{3-}eq kg N-eq mol N-eq	Indicator of the nutrient enrichment in freshwater, marine water, and terrestrial ecosystems (Helmes et al., 2012). Relates to more localised impacts.	• Excessive fertiliser application • Manure management • Livestock grazing • Irrigation practices • Pesticides and herbicides use • Soil erosion
Depletion of resources –Mineral resource scarcity/Fossil resource scarcity	Surplus Ore Potential (SOP)- kg Cu-eq Fossil Fuel Potential (FFP) kg oil-eq/MJ	Indicator of depletion of mineral and fossil fuel resource and reduced availability. Reflects intensity of direct and indirect energy use and the energy sources (Jungbluth and Frischknecht, 2010; Vieira et al., 2017)	• Farm machinery use • Transportation • Manufacturing of agricultural inputs • Climate control (heating, lighting, and cooling) • Irrigation
Acidification	kg mol H+	Indicator of the fate of a pollutant (create acidifying emissions) in the atmosphere and the soil (Roy et al., 2014), compounds that are precursors to acid rain.	• Fertiliser use • Acidic agrochemicals • Livestock waste • Soil acidification due to crops (legumes) uptake • Irrigation
Ecotoxicity-Fresh water Marine	(1, 4DCB-eq)	Indicator of date and effects of chemical emissions and impact on organisms of toxic substances emitted to the environment (van Zelm et al., 2013)	• Pesticides, herbicides, and fertiliser use • Fertiliser runoff • Improper management of livestock waste • Inefficient irrigation practices • Soil erosion
Human toxicity-Cancer and Non-cancer	CTUh	Impact on humans of toxic substances emitted to the environment.	• Pesticides and chemical fertiliser use • Livestock farming and animal waste • Foodborne pathogens • Mycotoxins • Heavy metal contamination
Land use	Ha or m^2 *ye annual cropland-eq	Indicator of the relative species loss/actual use of land and the changes in land use caused by a specific land use type (Curran et al., 2014; De Baan et al., 2013).	• Deforestation • Land clearing for planting (cropland expansion) and livestock management (grazing and pastureland expansion)
Water use	m^{3}·water-eq consumed	Indicator of the amount of water lost from a hydrological system, based on regionalized water scarcity factors. Some metrics measure unweighted water use (Hoekstra and Mekonnen, 2012).	• Irrigation • Water demand • Water quality and pollution • Soil quality • Water re-use and recycling
Ozone depletion	kg CFC-11-eq	Indicator that represents the amount of stratospheric ozone depletion caused by a substance (WMO, 2010)(WMO, 2010).	• Use of specific pesticides with ozone depleting substances
Photochemical ozone formation	kg NOx-eq to kg C_2H_2 eq	Indicators of emissions of gases that affect the creation of photochemical ozone in the lower atmosphere (van Zelm et al., 2016).	• Transportation • Fertiliser application • Agricultural machinery

case studies post-harvest cooling and drying are excluded from the case studies described here).

More generally, resource inputs and outputs need to reflect a comparable subsystem (e.g., fertiliser use per hectare compared with yield for a farm of similar size), regardless of whether multiple subsystems or a single representative case is presented. Therefore, identifying the deterministic or stoichiometric relationship between relevant input and output flows is a prerequisite for quantification. In this regard, yield is a crucial output and must be selected carefully, as input data are typically expressed in area terms. Key outputs—such as nitrogen losses (including direct field emissions and indirect emissions due to volatilisation and leaching) and fossil CO_2 emissions—are directly tied to inputs per hectare, reinforcing the need to ensure that yield estimates are representative values (Cederberg et al., 2013; Nemecek et al., 2015).

LCI development across livestock systems is arguably more complex due to the diversity of output types (e.g., live weight gain, animals sent to slaughter, milk, wool). While a simplified representation of herd or flock dynamics may be necessary in preliminary exercises, some accounting of these processes remains essential. It is therefore crucial to identify inputs that have a direct causal relationship with key outputs (see Fig. 4). For example, the quantity of nitrogen fertiliser applied embodies upstream burdens and directly influences both direct and indirect emissions at the field level. For AFPSs, the quantity and type of both compound and pasture feed determine not only embodied and direct land-use-related burdens, but also direct emissions from enteric fermentation.

Hence, it is important to ensure that causal inputs and outputs are endogenously related in the construction of inventories (i.e., N_2O emissions being directly linked to nitrogen inputs, rather than relying on external estimates such as generic emission rates per hectare).

Within the case study, a number of parameters are applied to standardise the relationship between inputs and fundamental outputs. Direct and indirect N_2O emissions from fertilisers and crop residues are calculated using the IPCC, 2019 Guidelines (Hergoualc'h et al., 2019; IPCC, 2019). Nitrate and phosphate leaching emissions are calculated with a detailed nitrogen balance equation based on Zampori and Pant (2019). For beef systems, herd composition is based on the herd dynamics- Global Livestock Environmental Assessment model (GLEAM) described by (MacLeod et al., 2018). Feed requirements based on energy requirements equations from the IPCC 2006/2019 guidelines (Volume 4, chapter 10)(IPCC, 2019; Joshi et al., 2006). Manure excretion and emissions are calculated with Tier 2 equations from IPCC, 2019 Guidelines (Gavrilova et al., 2019). For other livestock systems, existing datasets (comparable European Agri-footprint 6) were adapted to reflect UK conditions, such as was adapted the market in the UK, e.g. electricity grid mix, weight gain (broilers, sows, and piglets) (AHDB, 2022b; RSPCA, 2020). Manure excretion and emissions are calculated with Tier 2 equations from IPCC, 2019 Guidelines (Gavrilova et al., 2019). (Note that whilst treatment of on-field crop waste residues, feathers, fur, etc., are considered out of scope, manure and litter related impacts are included and straw is a co-product.)

2.7. Determine the relevant impact categories for meaningful life cycle impact assessment

The goal and scope of an LCA determine the impact assessment method relevant to the study (ISO, 2006a). When the goal is to gain insight into environmental hotspots, identify potential improvement opportunities, or compare alternative measures or product choices, it is essential that the assessment addresses the most meaningful environmental concerns aligned with the study's specific objectives. Each life cycle stage in an AFPS may lead to different environmental impacts, and relevant impact categories can vary depending on the process and methodological choices (Notarnicola et al., 2017).

Some studies may focus on a single indicator, such as global warming potential (GWP); however, a broader range of impact types is needed to identify specific trade-offs. Excluding impact categories that reflect burdens at different scales or ecosystem services undermines a systematic representation of impacts and reduces the usefulness of hotspot analysis. While GWP denotes a global burden, other impact categories—such as eutrophication—are more closely associated with localised impacts (ISO, 2006b) Indicators of resource appropriation (e.g., land, water, or energy consumption) relate to material use and can serve as important markers for both direct (e.g., on-farm cultivation area) and indirect (e.g., upstream arable land and energy) environmental stressors.

It is important to recognise that the specific units used are representative of potential burdens, not the impacts themselves. End-point indicators, which attempt to quantify manifest impacts (e.g., human health), are subject to higher levels of uncertainty compared to mid-point indicators (Huijbregts et al., 2017). For initial LCAs, it is advisable to focus on a limited number of high-priority impact categories rather than attempting to cover all possible categories. Additionally, the quantification of certain emissions (e.g., nitrogen-containing compounds, toxic substances) and impact categories (e.g., biodiversity, water scarcity, eutrophication) can be highly uncertain depending on the context.

Therefore, identifying key impact categories that are most relevant to both the local and global context—and significant to the AFPS under consideration—is crucial. For example, a regionalised water scarcity assessment based on country- or region-specific water scarcity factors is necessary to obtain meaningful insights into water use impacts (Kounina et al., 2013; Ponsioen and van der Werf, 2017). Common impact categories used in AFPSs include climate change, water use, land use, eutrophication, fossil resource use, and biodiversity (Notarnicola et al., 2017; Ponsioen and van der Werf, 2017). Table 4 presents relevant midpoint impact categories and associated activities in a typical AFPS.

A tailored impact assessment based on ReCiPe (Huijbregts et al., 2017) and IPCC, 2021 (IPCC, 2021) was used (employing SimaPro 9.5 software) to ensure the results of the considered case studies represent relevant impact categories. The specific impact categories considered are contained in Table 4, reflecting indicative global, local and resource use burdens. Specifically, these were: (i) GWP (kg CO_2-eq); (ii) marine eutrophication (kg N-eq); (iii) freshwater eutrophication (kg P-eq); (iv) terrestrial acidification (kg SO_2-eq); (v) water consumption (m^3); (vi) land use (ha*year); and (vii) fossil resources use (MJ). These were chosen to give a simplified overview of important trade-offs in preparation for an expanded assessment.

Other categories, such as ozone depletion and photochemical ozone formation, were excluded from the assessment due to irrelevance, whilst the toxicity impact was excluded due to the extreme uncertainty of the category. Studies such as Chen et al. (2021) demonstrated significant variability in differences in ecotoxicity (and other) results due to i) absolute emission values included in the inventory; ii) the inclusion of substances, and iii) the differences in the characterisation factor values of included substances.

2.8. General and case study methodological considerations

Whilst LCA methodology has evolved into a widely applied decision-making tool, it still has limitations (Guinée et al., 2011; Liu et al., 2024; Ponsioen and van der Werf, 2017; Weidema, 2014). Overcoming both theoretical and practical limitations is particularly crucial in the agri-food sector, where the inherent complexity of systems often exacerbates methodological challenges (Liu et al., 2024; Notarnicola et al., 2017). Fan et al. (2022) noted that research on the methodological and practical application of LCA in the agri-food sector remains limited compared to its use in industrial contexts. Liu et al. (2024) further highlighted that many traditional LCAs reported in the literature exhibit shortcomings in FU definition, system boundary setting, impact assessment method selection, and data quality control.

Transparency in methodological choices is essential to avoid misinterpretation and misuse of results (Gradin and Björklund, 2021). Table 5 presents illustrative studies and reviews in the agri-food sector with a methodological focus, while Table 6 summarises the key methodological aspects addressed in the case study.

2.9. Effective interpretation of results

Interpretation is a key stage in an initial LCA, where results are analysed to support effective decision-making and may inform further investigation through a full LCA. This aligns with ISO 14040, which emphasises the importance of interpretation in identifying areas for improvement (ISO, 2006a). Interpretation should be carried out systematically, potentially by comparing results with published studies in similar domains that use equivalent or comparable FUs, and by clearly stating recommendations and limitations to reduce uncertainty (Laurent et al., 2020).

Within a scoping study, it may be sufficient to compare process contributions to determine whether similar processes consistently contribute significantly across all impact categories, or whether certain processes appear significant only for specific categories—indicating potential hotspots. Although it is inadvisable to draw conclusions based on absolute values from scoping studies, understanding both relative contributions and absolute impacts is essential during interpretation, alongside recognising potential uncertainties (Zampori et al., 2016).

Many LCAs serve as reference points for policy support and decision-making; therefore, robust and transparent interpretation is critical (Sala et al., 2020). It is important to note that the types and magnitudes of potential hotspots will vary depending on the impact category, as well as the study's goal, scope, and assumptions. Consequently, any interpretation of hotspots must be cognisant of the study boundary to avoid misrepresentation, particularly when comparing with other case studies.

For example, Fig. 5a shows the relative contribution of various unit processes in the production chain of each animal-based food product

Table 5
Illustrative LCA studies (with methodological focus) in Agri-food production.

Reference	Agri-food product/System	Functional Unit	System boundary	Key issues/ relevance to scoping LCA/ limitations
(Williams et al., 2006)	Varied systems across England and Wales considering bread wheat, potatoes, oilseed rape, tomatoes, beef, pig meat, sheep meat, poultry meat, milk and eggs. Conventional and organic exemplars.	Varied by commodity, e.g. per tonne of carcass, per 20,000 eggs or per 10m^3 milk (approx 1 tonne dry matter)	Cradle to the farm gate.	Includes on-farm activities such as crop storage and drying or cooling for commodities like wheat and potatoes. Use of average data on items like fertiliser application. This can obscure meaningful differences between specific production methods, which is particularly relevant where different varieties are grown under the same system (e.g. horticulture). Argues for reframing climate burden as the 'carbon-nitrogen' footprint, given its prominence in the GWP of many arable and livestock systems. Emphasises the uncertainty associated with common mechanisms (such as the IPCC calculation).
(Nemecek et al., 2011)	Swiss arable cropping and forage production systems	Multiple, e.g land management function: per ha/year productive function: kg per dry matter and kg per Kilogram dry matter (DM) and MJ net energy for lactation (MJ NEL) financial function: return (expressed in CHF)	Cradle to the farm gate.	Organic systems have lower inputs but lower yields, shifting the outcome across units. Importance of modelling entire rotations and interactions at the cropping system level. Trade-offs between per-ha and per-product results complicate interpretation.
(Goglio et al., 2018)	Methodological framework on LCA crops and cropping systems	Argument for having both product-based and system-based FUs	Whole cropping system vs individual crop boundaries.	No single modelling approach fully captures the complex interactions in product LCA in cropping systems; therefore, it is recommended to adopt a cropping-system approach. Recommending to evaluate entire rotations and not just a single year of cultivation, at least including the crop following the legume Emphasise the need to use at least two FU (to capture the multifunctionality of the entire rotational cycle and product LCA) and the use of attributional and consequential LCA to capture different perspectives in the agronomic cycle.
(Poore and Nemecek, 2018)	Extensive meta-analysis harmonising LCA results considering global variation in 5 impact categories for 40 food groups.	Multiple, e.g. 100 g of protein for meat products 1 kg for vegetables and fruit 1 unit for alcohol	From input production to retail.	Some FUs are difficult to express with equal relevance across all categories (e.g. protein quality vs quantity). Implications for the role of producers vs consumers as the lowest-impact animal products exceed the average impacts of substitute vegetable proteins in terms of GHG emissions. Significant variability for staple crops, as 90th-percentile impacts are thrice the 10th-percentile impacts on all five indicators for crops such as the crops wheat, maize, and rice, upon which populations depend. Potentially relevant for initial LCAs as they offer opportunities to assess the position of results in the overall range.
(Costa et al., 2020)	Literature review on methodological aspects of legume crop rotation systems	Multiple as presented in the various literature	Multiple, ranging from single crop-cradle to farmgate to whole rotation multi-year boundaries	Crop rotation systems are inherently multifunctional (multiple crops and systems services); however, many studies are too narrow (product-only), missing the system-level dynamics.
(Majewski et al., 2020)	EU-based. multiple products including farm and fish products delivered to the consumer via short (e.g. pick your own) and long value (incl. logistics centre).	1 kg of purchased product	Cradle to grave (Farm to consumer, including waste), including farm, buildings, transport, and refrigeration leakage. Excludes road infrastructure.	The carbon footprint of 1 kg of produce provided via short supply chains ranged from 0.07 for internet sales to 1.54 kg CO_2/kg for 'pick your own'. For longer chains, this range was 0.24-0.4 kg CO_2/kg reflecting on-farm sales to intermediaries and sales to wholesalers, respectively. The 'pick your own' option reflected the highest impact per kg across all impact

(continued on next page)

Table 5 (*continued*)

Reference	Agri-food product/System	Functional Unit	System boundary	Key issues/ relevance to scoping LCA/ limitations
				categories (e.g. acidification, eutrophication, and other impacts) due to the much smaller quantities transported in single deliveries compared with longer chain trips. Impact of transit on food quality (e.g. amount usable by the consumer) may need additional analyses. Importance in presenting counterintuitive results in terms of the impact of elements like food miles.
(Detzel et al., 2022)	Comparison of European plant-based meat and dairy alternatives with conventional products	kg of product, 30 g of protein and 54 kcal of energy	Cradle to gate (raw material sourcing up to the final food products at the factory gate)	Vegetable-based meat alternative protein products combined with amaranth or buckwheat flour and a lentil protein-based vegetable milk alternative resulted in significantly lower climate burden than conventional systems (e.g., 132-240 g CO_2/ 100 g alternative product vs 625 and 957 g CO_2/ 100 g chicken product (for high performance and conventional chicken, respectively)). Similar relative difference, when results are expressed in protein terms but no consideration for protein quality. The majority of the impact of chicken is associated with direct land use change, reflecting the amount of soya used in animal feed.
(Goglio et al., 2023)	Consideration of criteria to be included in a harmonised LCA of livestock systems based on an expert workshop.	NA	Farm gate	Due to their complexity livestock systems' results suffer from low reliability and a high degree of variability. A research agenda for improving LCAs of livestock systems was identified to prioritise: i) Food, feed, fuel and biomaterial competition, crop-livestock interaction and the circular economy; ii) Biodiversity; iii) Animal welfare; iv) Nutrition; v) GHG emissions. Biodiversity presented the highest priority among respondents, which is considered very difficult to incorporate into LCA in a robust manner, especially within scoping or initial studies. Interestingly, the appropriateness of different co-product allocation schemes did not appear to be a priority.
(Porto Costa et al., 2023)	Comparison of conventional and alternative protein food value chains across 5 impact categories.	100 g serving of meat or plant protein product 1 L dairy and soya milk Refers to German conditions.	Up to the point of consumption but consider the impact of changes to the wider system.	Pea protein balls replace beef meatballs on a 1:1 mass basis; however, this would need to be confirmed based on nutrient quality. A nutrient-based LCA (nLCA) would add additional insights. Substituting beef meatballs with pea protein balls will save 2.42 kg CO_2e per 100 g serving. System perspective is important as the benefit of soya milk substitution is negated if surplus calf production is displaced, thereby needing a larger suckler herd. Resultant 'spared land' provides opportunities for additional on-farm carbon storage.
(Pérez et al., 2024)	Carbon footprint of organic berry production in Northern Spain.	Various: fruit mass, 1 kg of fruit, 1 ha of land, 1 € of farm-gate price, and nutritional quality of fruits.	Farm gate	The relative performance of commodities with the highest and lowest carbon footprint was not affected by FU choice, whereas the relative ranking of other fruits was determined by FU choice. This means that in seeking to reduce impact, the ranking of commodity choice based on mass or land use will differ from the ranking of commodities based on an economic or nutritional metric.

(*continued on next page*)

Table 5 (*continued*)

Reference	Agri-food product/System	Functional Unit	System boundary	Key issues/ relevance to scoping LCA/ limitations
(Kiehbadroudinezhad et al., 2025)	Review of published LCA results to explore implications for global agriculture of renewable energy uptake.	Sectoral level results presented, but various FUs expressed based on literature presented, e.g. 1 kg of tomato production or 1 kWh of energy delivered to the grid.	Global sectoral level	Each renewable energy source embodies different trade-offs amid decarbonisation of the global agricultural sector, and a focus on GWP may ignore these important impacts when applied at scale, e.g. land use demands and lithium for solar. In particular, less attention is paid to human health impacts, specifically those associated with material production to disposal. Renewable energy and appropriate technologies are necessary for the decarbonisation of processes stage, but low-impact options (e.g. solar drying) are gaining exposure. When comparing energy outcomes, the choice of FU is important for biofuel as biodiesel may have a lower calorific value, mean comparison against conventional diesel is misrepresentative. Opportunities for co-benefits via emerging technologies, e.g. enhanced bio-sequestration in soils and above ground carbon sequestration within sheep agrivoltaics systems.

(ABFP) from cradle to farmgate. This excludes post-farmgate transport and should ideally be compared with studies using similar boundaries. The process contributions to the production of 1 kg of each plant-based food product (PBFP) at the farmgate are shown in Fig. 5b. In ABFPs, feed production dominates most impact categories—such as marine and freshwater eutrophication (due to nutrient losses via runoff and leaching) and land use (due to the agricultural area required for feed production).

However, direct comparisons of environmental impacts across commodities are not always appropriate due to wide variations in production systems and associated intangible benefits. For instance, beef production has a significantly higher GWP compared to pig production (per kg liveweight; see Table 10), but beef farming can utilise grassland that may be unsuitable for producing crops edible by humans or other monogastric animals. This requires contextualisation: while these systems yield different slaughter by-products (necessitating allocation), the scale of impact differences suggests that some relationships may persist beyond the farm level and into consumer-level assessments. Using economic allocation, Blonk et al. (2022) estimate a similar GWP estimate for 1 kg of Danish Pig and Irish beef at the farm, consistent with the estimates presented in this study when expressed in liveweight (Table 10).

Similarly, for the arable crops considered (potatoes and winter wheat), eutrophication is heavily influenced by nitrogen losses at the field level (Fig. 5a). GWP in arable production does not appear to be dominated by a single category. In farm-level assessments, most GHG emissions (GWP100) tend to arise from direct livestock emissions—-primarily enteric methane (>52%)—and, depending on housing, manure management (>6%). Farm infrastructure is particularly relevant when considering fossil resource depletion. In PBFP systems, field emissions dominate most impact categories for potato and wheat production, whereas heat production is the major contributor in protected tomato systems.

These insights, derived from hotspot identification, can serve as precursors to more in-depth studies. A key consideration when comparing results across and within commodities is the identification of trade-offs. An agricultural commodity may exhibit a lower impact in one burden type but a higher impact in another. Trade-offs may also occur within the same impact type and boundary—for example, reduced energy use at one value chain stage may result in increased impact elsewhere.

Prior consideration of potential trade-offs—whether inter- or intra-commodity, or inter- or intra-impact type—will assist in interpretation. Most commodities demonstrate at least one process contributing ≥50% of the impact across several categories, such as animal feed production in Fig. 5a. In such cases, it is equally important to identify impact categories where these processes do not significantly influence overall burdens—for example, terrestrial acidification in poultry systems.

The designation of specific processes should be determined by the practitioner based on the flows of individual environmental burdens that share characteristics. It is important to distinguish direct emissions at the farm level or, at a minimum, use categories that are sufficiently distinct to inform tangible management decisions (e.g., distinguishing energy-related and biogenic sources of GWP). Additionally, it is important to identify processes for which no clear hotspot exists, necessitating multiple interventions—such as GWP in potato production.

3. Discussion

3.1. Insights from presented case studies

Given the research focus and growing demand for LCA in AFPSs, many publications report conclusions based on initial LCAs. On the one hand, accurate interpretation of results requires subject-matter expertise, as LCA practitioners may sometimes draw overly strong or misleading conclusions (Notarnicola et al., 2017; Notarnicola et al., 2012). On the other hand, poor assumptions and data choices throughout the analysis can complicate interpretation, as hotspots identified at this stage may not accurately reflect the overall burden—particularly when absolute values are not considered.

For example, findings from the presented case study suggest that feed production is the main contributor to environmental burdens in ABFPs, field emissions dominate arable crop impacts, and heat production is the primary driver in protected crop production. However, the use of lower-

Table 6
A summary of methodology in the food basket case study.

Commodity	Beef (from beef cattle)	Pork	Chicken meat (from broilers)	Eggs (from layers)	Ware potatoes	Spring wheat grains	Tomatoes-in a field polytunnel	Tomatoes in a heated greenhouse
Functional unit	1 kg liveweight at farm at slaughter-ready age			1 kg fresh egg at the farm gate	1 kg fresh potatoes at the field gate, ready for chilling/ storage	1 kg fresh dry winter wheat grain at the field gate, ready for storage	1 kg fresh loose tomatoes at the field gate, ready for chilling	
System type	Intensive lowland spring-calving, grazing	Intensive Indoor systems			Conventional arable systems		Unheated polytunnels	Heated glasshouses
System boundary	Cradle to farm- gate				Cradle to field- gate			
Activities included	Manure management emissions (CH_4, N_2O, NH_3 etc.) Livestock and soil emissions (e.g., enteric fermentation, field emissions (N_2O)) Land occupation and grass/silage yield (for cattle) Feed production (grass, silage and compound feed) Housing system (electricity use) Transport of inputs Diesel use				Manure application emissions (production excluded - cut-off) Chemical and lime fertiliser production and application emissions Pesticides production and application emissions (i.e. N_2O, NH_3, leachate etc.) Seed/start material production Capital goods (e.g., infrastructure, trellis system, plastic tunnel, glasshouse) Energy use for cultivation (farm diesel production and combustion, electricity production for lighting and pumping, natural gas production and combustion for glasshouse heating) Irrigation water input Land occupation and crop yield			
Allocation	See Table 2, section 2							
Data sources	See Tables 1–8S supporting information- B for LCIs							
Temporal scope	2021/2022	2021/ 2022	2021/2022	2021/ 2022	2018-2020	2021/2022	2019/2020	2021/2022
Geographic scope	UK	UK	UK and Netherlands	UK	UK	UK	UK (and Spain)	UK
Impact categories	See Table 4, section 2							
Key assumptions influencing the results	Grass-based system Herd composition Digestibility of feed The fraction of the assumed split between indoor and outdoor management.	For pig systems: feed conversion ratio; litter size and mortality; feed composition and sourcing (Note feed also determines manure quantities); efficiency of manure management. For poultry: feed conversion ratio; feed composition and sourcing (particularly the % allocated to regions such as Argentina and Brazil, with high deforestation potential); efficiency of manure management.			Yield, fertiliser use, and energy required for ploughing and harvesting. This is particularly relevant for potatoes given the sowing depth. For wheat, the measured moisture content of spring wheat harvested in the study year meant that limited additional drying was needed.		Levels of fertiliser applied and leachate rates; minimal yield loss; functional life of plastic material.	Yield systems; level of heating; energy needed.
Limitations and complications	For pork and poultry, a conservative mortality and feed conversion estimate. Sourcing of soya was based on trade statistics, which might not be representative for the market segments. For the beef systems, a simplified herd dynamic and feed requirement were used, whereby the approximate breakdown of grass and compound feed is suitable for lowland systems with adequate pasture quality (e.g., days outdoors).				The national average yield is used, but this will vary significantly. Soil quality and weather conditions will influence ploughing and harvesting fuel use. For wheat, no additional drying was included due to the dry harvest period of the study year. This, therefore, is a snapshot which not representative of general conditions. However, hot night temperatures may result in external cooling needed. This has not been taken into account may become more widespread in future years. Wheat straw is assumed to be baled. Other practices of crop residue management (including residues left in the field) were not considered and were deemed minimal at the field level.		Coarse consideration of seasonal variability (e.g. heating demand) for indoor systems. No external CO_2 inputs assumed. Yield data from the equivalent Dutch system is used as specific data on current UK heated systems, which is unavailable. Additional chilling was considered post-harvest. Larger installations will have on-farm chilling.	
Uncertainty	See Table 8, section 3							

quality feed, food waste, or domestic grains for animal feed, and the utilisation of grazing land unsuitable for arable crops in beef farming, could challenge this interpretation—especially when compared to more conventional or intensive systems. Nevertheless, such comparisons remain valid, as noted by Ponsioen and van der Werf (2017), if the goal is to identify which production systems contribute most to impacts at the population (overall consumption) level. In this context, any hotspot identified should be interpreted as a preliminary indicator, pending confirmation through a full LCA with more coherent data and sensitivity analysis.

Initial LCAs often rely on generalised secondary data that may not account for regional practices, climate, or soil conditions, potentially leading to misleading interpretations when applying results to specific regions. Generalising results from a specific context to a broader scale may fail to capture the full range of environmental impacts and limit the applicability of findings. Attributional LCA—which describes the environmental burdens directly associated with the life cycle of a product using average data from existing systems—is commonly used to represent sectors or regions. However, this approach can be problematic when averages are applied in contexts requiring greater specificity. For

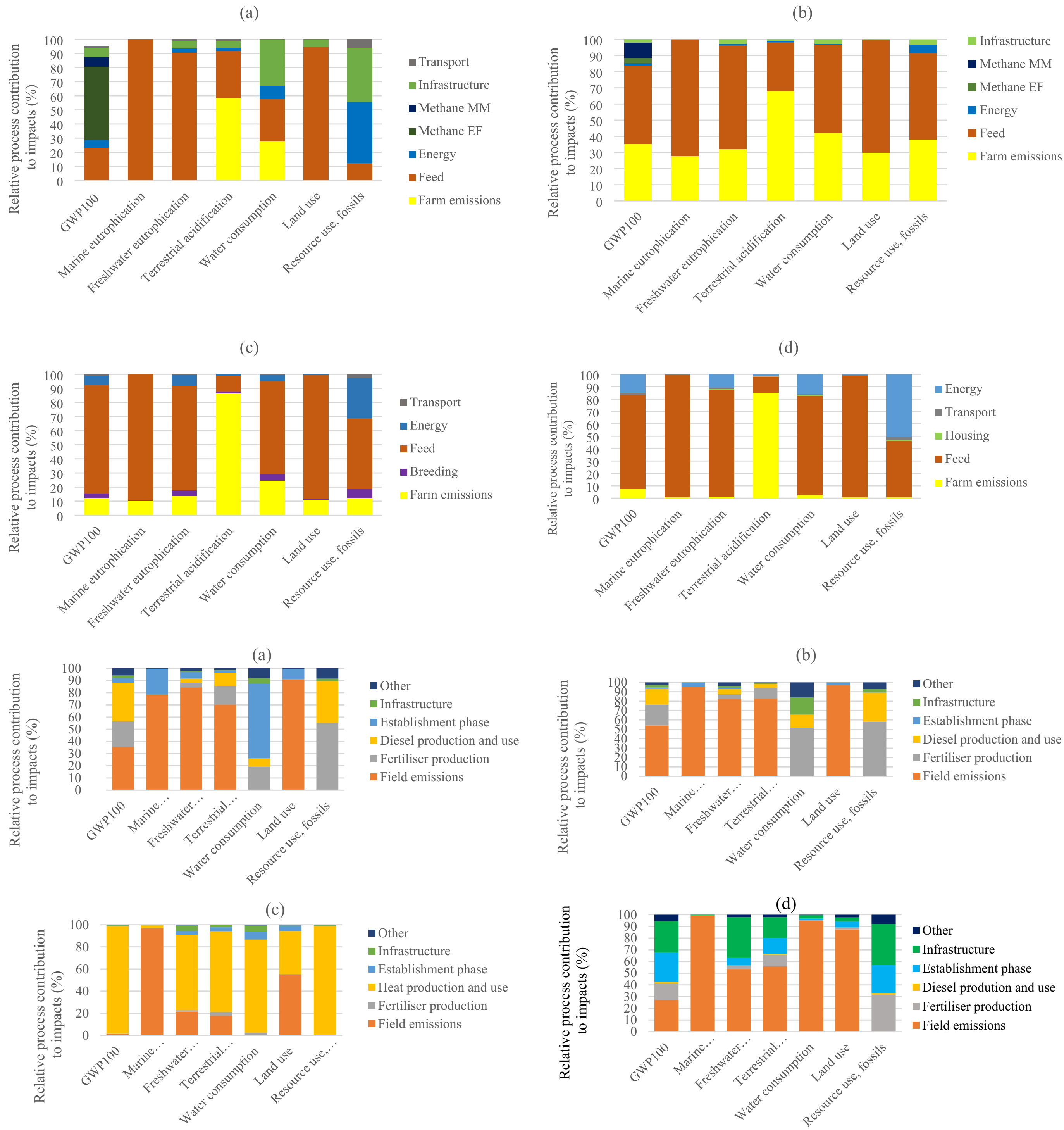


Fig. 5. a: Relative contributions of major unit processes from cradle to farmgate to produce 1 kg liveweight leaving the farm in a representative (a), lowland grazing beef; (b), intensive indoor pig; (c), intensive broiler; and (d), eggs in a representative enriched cages layer; production system in the UK under each of the seven impact categories (note direct livestock includes emissions due to manure management and field emissions). A combined impact assessment based on ReCiPe (Huijbregts et al., 2017) and IPCC, 2021 (IPCC, 2021) was used (impact assessment using Simapro 9.5 software).
b: Relative contributions of major unit processes from cradle to farmgate to produce 1 kg of (a), freshly harvested potatoes; (b), winter wheat; (c), glasshouse produced tomatoes in, representative production systems in the UK; and (d), unheated polytunnel produced tomatoes in a representative production system in Spain under each of the seven impact categories. A combined impact assessment based on ReCiPe (Huijbregts et al., 2017) and IPCC, 2021 (IPCC, 2021) was used (impact assessment using Simapro 9.5 software).

instance, the average annual potato yield in the UK (2018–2020) is 46 tonnes/ha and was used in the LCI for the exemplar case study. Yet regional yields vary from 39 to 49 tonnes/ha due to differences in fertiliser use, irrigation, cultivar selection, and agronomic efficiency, all of which influence local environmental impacts (AHDB, 2021; PotatoPro, 2023). A single average value may not be representative: approximately half of UK potato crops are irrigated, but using half the irrigation volume as a proxy may misrepresent actual conditions. Therefore, consideration

Table 7
Comparison of total burden at the field gate per production of 1 kg of fresh loose tomatoes produced in the UK and Spain.

Impact category	Unit	UK (heated glasshouse)	Spain (field polytunnel)
GWP	kg CO_2-eq	1.6859	0.1927
Marine eutrophication	kg N eq	0.0003	0.0011
Freshwater eutrophication	kg P eq	0.0001	0.0003
Terrestrial acidification	kg SO_2 eq	0.0019	0.0018
Water consumption	m3	0.0015	0.0273
Land use	m2a crop eq	0.0236	0.0485

Table 8
Example data requirements and potential data sources for a typical AFPS.

Data requirement			Potential data sources
Specification		Examples and Remarks	
Agricultural practices	LUC/ Crop rotation/ Irrigation/ Fertilisers/ Pesticides/ Machinery usage	• The amount, rate, and type of fertiliser applied per hectare. • The type and quantity of pesticides used. • Machinery use efficiency, time, during planting, harvesting, etc. • Time spent grazing vs indoors.	• National agricultural statistics, agents (such as Defra, AHDB for the UK).
Yield data	Crop/pasture yield	• Yield per hectare which includes information on average yields, variations due to different varieties, cultivation practices, and the factors influencing yield fluctuations. • Depending on the crop may be expressed as dried weight. • Grass yield and quality (for livestock systems). • Helps in understanding the efficiency of resource utilization and impacts on land use. • It is important to be aware of whether the yield includes on farm losses or not.	• Agricultural research institutes and universities. • Agricultural advisory services or extensions. • Scientific literature.
Herd Dynamics and outputs	Calving/lambing rates, culling, and growth rates.	• Herd composition by number (i.e., livestock density), gender, maturity, etc.) • Percentage mortality, weaned, replacement, saleable animals • Growth rates by animal type. • Milk and wool yield per animal.	• Agricultural advisory services or extensions. • Livestock Management Software.
Land use data	Land area/ Soil type/ Land management practices/ Potential impact on biodiversity	• Data on crop rotation. • Depending on the setting it may be important to consider if LUC occurred prior to use. • Pasture quality.	• Geospatial data sources. • Scientific literature.
Water use data	Water sources/ Irrigation techniques/Water use efficiency	• Use of ground water or surface water. • Amount of water used per hectare. • Animal water needs.	• Water management authorities. • Farm audits. • Other literatures.
Energy data	Energy use for machinery operation/ heating/ lightning/ Drying/ Irrigation/ Transportation	• Specific to the process, use of farm machinery on the field (e.g., fertiliser application), which may need to consider number of applications. • May be presented on a per area or per output basis. • Often presented as aggregate (e.g., L/ha) estimate but, if possible, it is important to distinguish impactful processes, ploughing vs fertiliser use. • Some elements (like drying demand) will depend on moisture content.	• National authorities. • Farm energy audits
Transportation	Mode/vehicle type/function	• Available transport links depending on the function/distances/viability. • Whether it is an inter continent or international and/or regional transportation.	• Geospatial data sources.
Material Use	Capital or consumable equipment	• Seedings, pesticides, fertiliser. • Plastic bags trays, soil, compost or straw, boxes and tray. • Results will need to consider usage intensity.	• LCA databases (Eco invent, Agri-footprint, WFLDB, etc.). • Farm Surveys.
Emission factors	Fuel combustion/ Fertilizer application/ LUC	• GHG emission from fuel combustion. • N-N_2O emission factors from N-based fertilisers that distinguish fertiliser type. • Emissions related to LUC. • Emissions (NH_3, N_2O, and CH_4) due to manure management. • Leachate and run-off estimates.	• IPCC guidelines. • National emission guidelines. • LCA databases (Eco invent, Agri-footprint, WFLDB, etc.). • Scientific literature.
Market data	Crop prices/Market demand	• Need to ensure the prices used for allocation across co-products are consistent and appropriate to the boundary (e.g., Farmgate vs market vs wholesale prices).	• Market research reports. • International data bases (FAO, USDA, etc.)
Data on waste and by-product	Waste generated (amount/rate) during the crop production, etc.	• This includes quantifying the amount of waste generated and its fate (for e.g., unused crops and post-harvest losses). • In mixed livestock, it is necessary to define the treatment method for manure (e.g., how it is stored on farm).	• Farm surveys. • Local waste management authorities. • Scientific literature.

Remarks: Defra, Department of Environment, Food and Rural Affairs; AHDB, Agriculture and Horticulture Development Board; WLDB, World Food LCA Database; IPCC, Intergovernmental Panel on Climate Change; FAO, Food and Agriculture Organization; USDA, United States Department of Agriculture.

of regional exemplars is also necessary to provide context. This is also a simplification, as yield will respond to irrigation and options such as the use of recycled and harvested water may reduce its resource burden. Further, the omission of residue and soil dynamics is one of the major limitations of initial LCA. Integrating soil- and agroecosystem modelling can reflect these dynamic interactions rather than ignoring residues or treating them as waste or outside the boundary (Siol et al., 2025). This, however, is likely due to a lack of site-specific data in the initial assessments, as generic databases often miss these flows.

However, a potential drawback of conducting a more thorough or

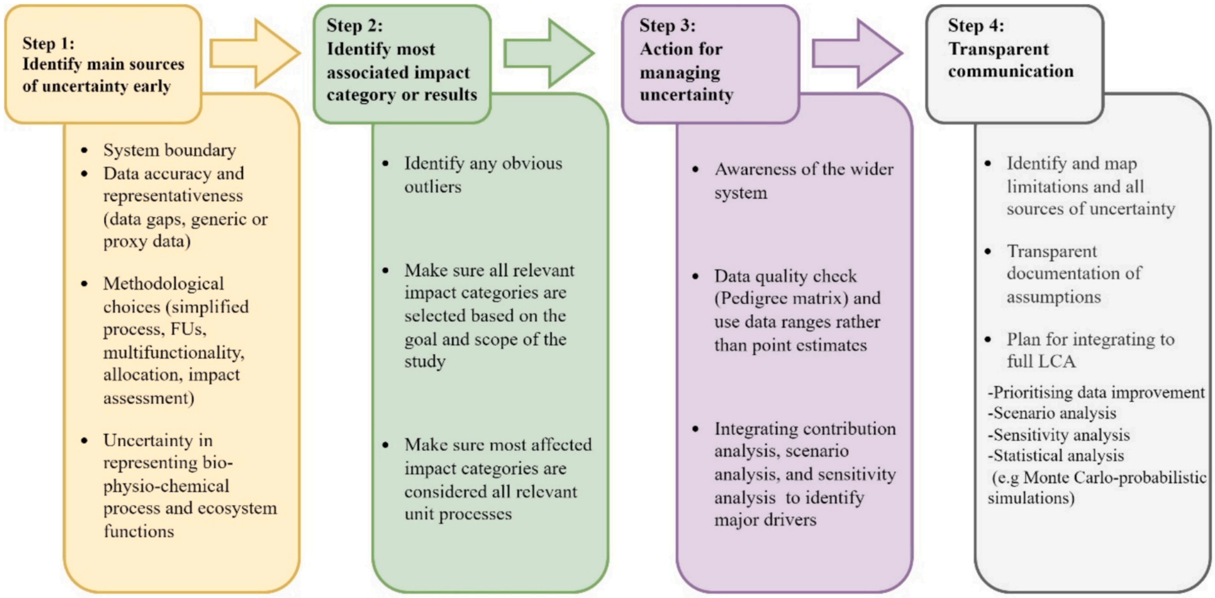


Fig. 6. A structured framework of operational guidance on managing uncertainty in an initial LCA (designed by adopting insights from Bamber et al. (2020); Barahmand and Eikeland (2022); Helmes et al. (2022)

detailed LCA is that it may reveal greater environmental impacts, which can discourage businesses from investing in the data collection necessary to support such models (Blonk et al., 2010; Cederberg et al., 2013; Ponsioen and van der Werf, 2017). Initial LCAs—particularly in the agri-food sector—often focus on immediate impacts, while indirect impacts such as LUC, deforestation, and socio-economic effects may be overlooked or only partially considered due to limited data availability or simplified methodologies (McAuliffe et al., 2023a). These limitations are compounded by the difficulty of communicating complex results to non-expert audiences (Heijungs, 2021; Soust-Verdaguer et al., 2016)

Regardless of the sophistication of the impact assessment model, LCA does not adequately capture essential information such as the volume of the receiving compartment or the duration of exposure—factors critical to assessing the true toxicity of a chemical released into the environment (Notarnicola et al., 2017). Therefore, it is essential to interpret such impacts with caution, avoiding both underestimation and over-estimation, to ensure the feasibility and credibility of an initial LCA.

In the case studies presented here, the farm or field gate boundary was selected as the initial point of analysis, as it represents a common stage across most products and, in many cases, reflects the most impactful phase. For example, comparing UK tomatoes grown under heated glasshouse conditions with Spanish polytunnel systems (such as those found in Almería) demonstrates that the latter may have a lower GWP, but this comes at the cost of increased risks of eutrophication and water scarcity (see Table 7 and Fig. 1S in supporting materials-A). These impacts are driven by fertiliser use and associated nutrient losses through leaching (Zampori and Pant, 2019), as well as evaporative losses from irrigation and in-situ water stress.

In this case, the international comparison highlights how differences in production systems interact with climate-related drivers of water stress, reinforcing the importance of contextualising LCA results within both environmental and regional frameworks.

As mentioned above, the inclusion of post-farm/post-harvest activities can, in some cases, contradict or ameliorate the apparent differences in impact between alternative products at the farm/field gate. However, this is context-specific and depends on the processes that contribute to each individual impact type. By way of *example* if the study boundary for tomatoes is *extended* to include post-harvest processes such as chilled storage, processing at a depot and chilled transport (overland truck and ferry in the case of Spanish Tomatoes), terminating in a UK depot, the relative difference in GWP is reduced (1.17 kg CO_2e/chilled kg vs 2.2 kg CO_2e/Chilled kg for Spanish and UK tomatoes at a UK depot) however the trade-off between GWP and more localised eutrophication impacts remain, as the latter are heavily concentrated in the farm stage (Fig. 1S, supporting materials-A).

3.2. Managing data requirements

Lack of data or reliance on incomplete or outdated data can lead to uncertainties and affect the accuracy of the assessment. Table 8 presents guidance on the qualitative and quantitative data requirements of a representative AFPS. It should be specified that the data requirements in Table 8 do not distinguish between a full and initial study, but their application in either may differ. For example, whilst any grazing livestock LCA will need to account for co-products, a complete herd dynamics model may not be necessary for an initial study.

Similarly, databases, such as Agri-footprint (Blonk et al., 2023), AGRIBALYSE (Colomb et al., 2015; P. Koch, 2015), Word Food LCA Database (WFLDB) (WFLDB, 2022), GaBi Food & Feed (Burhan, 2018) and Ecoinvent (B.P. Weidema, 2013; Burhan, 2018), provide an extensive resource of agricultural background inventory data. However, it is recommended to carefully weigh the advantages and limitations of using existing databases (see Table 3S-supporting materials-A) and to complement them with additional data sources, expert knowledge, and site-specific information to achieve a more comprehensive understanding. For example, when the contribution of a particular input in a database is minimal (e.g., <0.1%), it may be acceptable to use a proxy. However, for inputs with more substantial contributions, more specific values should be sought before relying on proxies.

Similarly, when one database includes an essential input or flow that another omits, consolidating data from both sources may yield a more complete dataset than either database alone. A comparable approach can be applied to identify missing flows by consulting alternative resources. This strategy helps ensure that the LCI reflects the full scope of relevant processes and avoids underrepresentation of key environmental burdens

Corrado et al. (2018) demonstrated that datasets across various databases used to model the same commodity often exhibit substantial methodological variation, leading to significant differences in impact assessment results. While the need for region-specific data may depend

Table 9
Potential sources of error in developing LCI of the exemplar case studies.

Sources of uncertainty	Commodity/ FU at farm gate							Most associated impact category	Data/methodological options for expansion.
	Livestock (liveweight)		layers		Field crops		Protected greenhouse crops		
	Broiler	Pig	Beef	Chicken eggs	Potatoes	Wheat	Tomatoes		
Unrepresentative feed composition (energy and protein level)	*	*	*	*				Climate Change Fossil resources use Eutrophication	Identify a more representative feed mix, including sourcing
Unrepresentative feed sourcing (geographic origin/ quality aspects)	*	*	*	*				Climate Change LUC Eutrophication	Scenario/sensitivity analysis with different potential feed compositions with different sourcing.
Using generic emission factors - Manure management without considering soil type, climate, or management practices	*	*	*	*				Almost all impact categories	Methodological development (e.g. define more specific/targeted emission factors based on soil conditions, e.g. IPCC tier 2/3, taking into consideration specific manure management).
-Over and underestimating feed conversion (due to poor data on feed intake or variation in animal growth rates)	*	*	*	*				Climate Change Eutrophication Acidification	Consider the sensitivity of feed conversion ratio.
-Underestimating Nitrogen loading from litter/manure	*	*	*	*				As above	Both stocking rates and feed quality will determine the overall litter production and accumulation. Check for consistency with IPCC GHG emission estimation methodology. Use production specific (e.g. indoor vs outdoor cattle) waste and N loading methods.
-Housing duration	*	*	*						
-Simplifying LUC impacts -Misinterpretation of the emissions from deforestation or pasture conversion due to grazing or feed production	*	*	*	*	*	*		LUC Climate Change	Check assumptions on embodied LUC change emissions (e.g. what changes in land use were assumed, over what time frame).
-Underestimation of soil organic carbon	*	*	*	*	*	*		As above	Acknowledging residue management and soil dynamics helps avoid blind spots. Include sensitivities w/wo impact of soil carbon accumulation based on literature assessment for appropriate regions. For carbon loss assess if region of specific crop cultivation has significant carbon content (e.g. peaty soils). Consult the IPCC Tier 1 methods for modelling soil carbon (i.e. multiply reference soil carbon stocks and stock change factors related to land use, management actions and inputs) (IPCC, 2019).
-Underestimation and misinterpretation of residue management					*	*			
-Using CH_4 emission factors for enteric fermentation that do not account for changes in feed formulation.			*					As above	Consider adapting existing method IPCC tier 2 calculation methodology including specific estimates on digestibility (IPCC, 2019).
-Underestimation of soil N_2O emissions	*	*	*	*	*	*		Climate Change Eutrophication Acidification	Methodological development (e.g. define more specific/targeted emission factors based on soil conditions, crop rotation, residue management, etc. IPCC tier 2/3)
Use of generic secondary data -Over and underestimation of crop yield					*	*	*	For all impact categories	Always use the national statistics. Unless representing a specific region.
-Fertiliser and pesticide use (type and application rate) (feed production in livestock and layers; main crop production)	*	*	*	*	*	*	*	Climate Change Eutrophication	As above
-Combined use of (e.g. regional) average data for fertilizer and pesticides production while using site-specific data for crop yields						*	*	For all impact categories	Consider producing additional exemplars reflecting sub value chains for which highly granular estimates of inputs and yields are available.

(*continued on next page*)

Table 9 (*continued*)

Sources of uncertainty	Commodity/ FU at farm gate							Most associated impact category	Data/methodological options for expansion.
	Livestock (liveweight)		layers		Field crops		Protected greenhouse crops		
	Broiler	Pig	Beef	Chicken eggs	Potatoes	Wheat	Tomatoes		
Generic data on energy use -Under and over estimation of farm machinery use and other agronomic practices	*	*	*	*	*	*	*	Climate Change	More detailed study identifying if energy constitutes a greater hotspot (e.g. in protected tomato cultivation).
-Using outdated energy profiles for greenhouse operations (e.g. ignoring LED lighting)							*	Fossil resources use	
-Assuming uniform emissions for greenhouse energy use (without accounting for regional energy grids)									
-Ignoring impact of other agricultural practice (e.g. crop rotation, drying etc).					*	*		For all impact categories	Check seasonal/regional variation[a]
Allocation								As above	More detailed study identifying it a greater hotspot The use of sub-processes
-Misrepresenting changes of herd dynamics (growth rates, digestion efficiency, reproductive rates within a herd), mass balance	*	*	*	*					
-Misallocating burdens between co-products (meat, milk, manure, grain, waste, straw)	*	*	*	*	*	*		As above	Compare results under different allocation schemes to determine sensitivity to allocation scheme, particularly in relation to commodity of focus. Ensure waste and non-waste co-products are distinguished.

[a] In the case study year for wheat, the winter wheat harvest period for the study year experienced temperature meant that wheat moisture level necessitated minimal on farm drying, this cannot be generalised. This also provides equanimity with tomato systems that did not assume additional cooling.

on methodological choices (Castellani et al., 2017; Gheewala et al., 2020; Notarnicola et al., 2017), many databases still lack detailed and regionally representative datasets (Notarnicola et al., 2017).

To address this, a pedigree matrix can be employed to assess the actual quality of collected data—evaluating dimensions such as reliability, completeness, temporal, geographical, and technological representativeness. This tool is particularly useful during the scoping phase and can also help define specific data quality goals for a full LCA (see Table 4S, supporting materials-A).

3.3. Managing uncertainty

In LCA, the predominant sources of uncertainty include parameter uncertainty, data variability and quality, scenario uncertainty, diverse methodological choices, model uncertainty, and impact assessment frameworks (Bamber et al., 2020; Barahmand and Eikeland, 2022). Initial LCAs often rely on incomplete or low-quality data due to time and resource constraints, which can result in high levels of uncertainty and potentially misleading conclusions (Gradin and Björklund, 2021; Kiemel et al., 2022; Pernollet et al., 2017).

While conducting a formal uncertainty analysis is often beyond the scope of an initial LCA, some accounting of uncertainty is necessary to meet study objectives and guide iterative improvements. Fig. 6 presents a structured framework for identifying and managing uncertainties in an initial LCA.

Although qualitative evaluation of uncertainty is possible—based on data quality, model assumptions, and system boundaries—rigorous uncertainty analysis requires quantifying uncertainty for each input (e.g., using ranges and probability distributions) and reflecting this in the impact estimation (Bamber et al., 2020). This process demands both methodological expertise and a comprehensive understanding of the wider system under consideration (Notarnicola et al., 2017).

Table 9 presents potential areas of uncertainty and the impact categories affected in the exemplar case studies. Consequently, the use of default or proxy information is often necessary to address data gaps. However, default data—such as yield factors based on average agronomic parameters or historical data—may not provide reliable estimates due to variability in management practices and performance at regional levels.

Depending on the objectives of a full LCA, a detailed uncertainty analysis, including sensitivity and scenario analyses, would offer more robust insights (ISO, 2006b). When sufficient data are available, the uncertainty of inputs and outputs can be assessed by comparing annual average data collected over a defined multi-year period (Cerutti et al., 2014), or by comparing estimates across regions.

For variables such as yield, conventional field-level sampling may produce inaccurate estimates at a given time or yield values may only be representative within a limited temporal window. As such, conducting an adequate uncertainty analysis can itself increase data requirements. Analogous to the pedigree matrix, uncertainty due to data limitations can be assessed across several dimensions, such as reliability, completeness, temporal, geographical, and technological representativeness, using a scoring scale (typically from 1 to 5) (Igos et al., 2019) as in Table 4S (in supporting materials-A). These scores can then be translated into uncertainty factors, which help quantify the confidence level associated with each input and output in the LCI. This approach supports more transparent interpretation and prioritisation of data improvement efforts, particularly in transitioning from an initial to a full LCA. A quantitative, rather than qualitative uncertainty analysis, leads to a more transparent increase in confidence in the LCA findings and adds considerable value to the decision-making process, going beyond merely indicating the confidence decision makers have in the results (Guo and Murphy, 2012).

Whilst uncertainty analysis seeks to reflect a range of possible

Table 10
Comparison of preliminary UK LCA results (GWP- kg CO_2 eq) with literature to understand the possible uncertainty in the study.

Commodity	GWP (kg CO_2 eq/kg) This study	GWP (kg CO_2 eq/kg) Examples from literature
Fresh Potato	0.08	0.21 Italian systems for early potatoes (Timpanaro et al., 2021) but these studies have a lower yield and include storage. 0.23 for UK potatoes, including on-farm storage (Williams et al., 2006). 0.115 Potatoes at a farm in the UK (Blonk et al., 2022). 0.09 US systems taking the post-farm losses and % of farm stage into account (Parajuli et al., 2021).
Winter Wheat	0.36	0.2-0.5 in Spanish systems (González-García et al., 2023). 0.31-0.4 in Polish winter wheat ((Holka and Bieńkowski, 2020). 0.8 for UK bread wheat (including on-farm drying) (Williams et al., 2006).
Fresh Tomatoes	1.68	0.78-2.0 Dutch indoor tomatoes, (Antón et al., 2012). 1.08 Dutch indoor heated hydroponic tomatoes (WFLDB, 2022).
Eggs	3.43	2.9-3.45 UK, (Leinonen et al., 2012a). 2.75 Dutch systems (Blonk et al., 2022).
Live Broiler	2.80	3.1 Standard UK systems adapted from (Leinonen et al., 2012b) converted to liveweight, assuming a 70% carcass yield. 2.4 liveweight in Dutch Systems (Blonk et al., 2022).
Live Pig	2.74	2.2 Live weight, UK, (Rudolph et al., 2018). 2.4 Live weight, Denmark, (Nguyen et al., 2011). 2.95 Live weight, UK (Blonk et al., 2022).
Live Beef	17.7	8-14 Live weight for Irish systems (Herron et al., 2021). 15 Live weight, (Blonk et al., 2022). 8 adapted from (Williams et al., 2006) converted to liveweight assuming a 50% carcass yield.

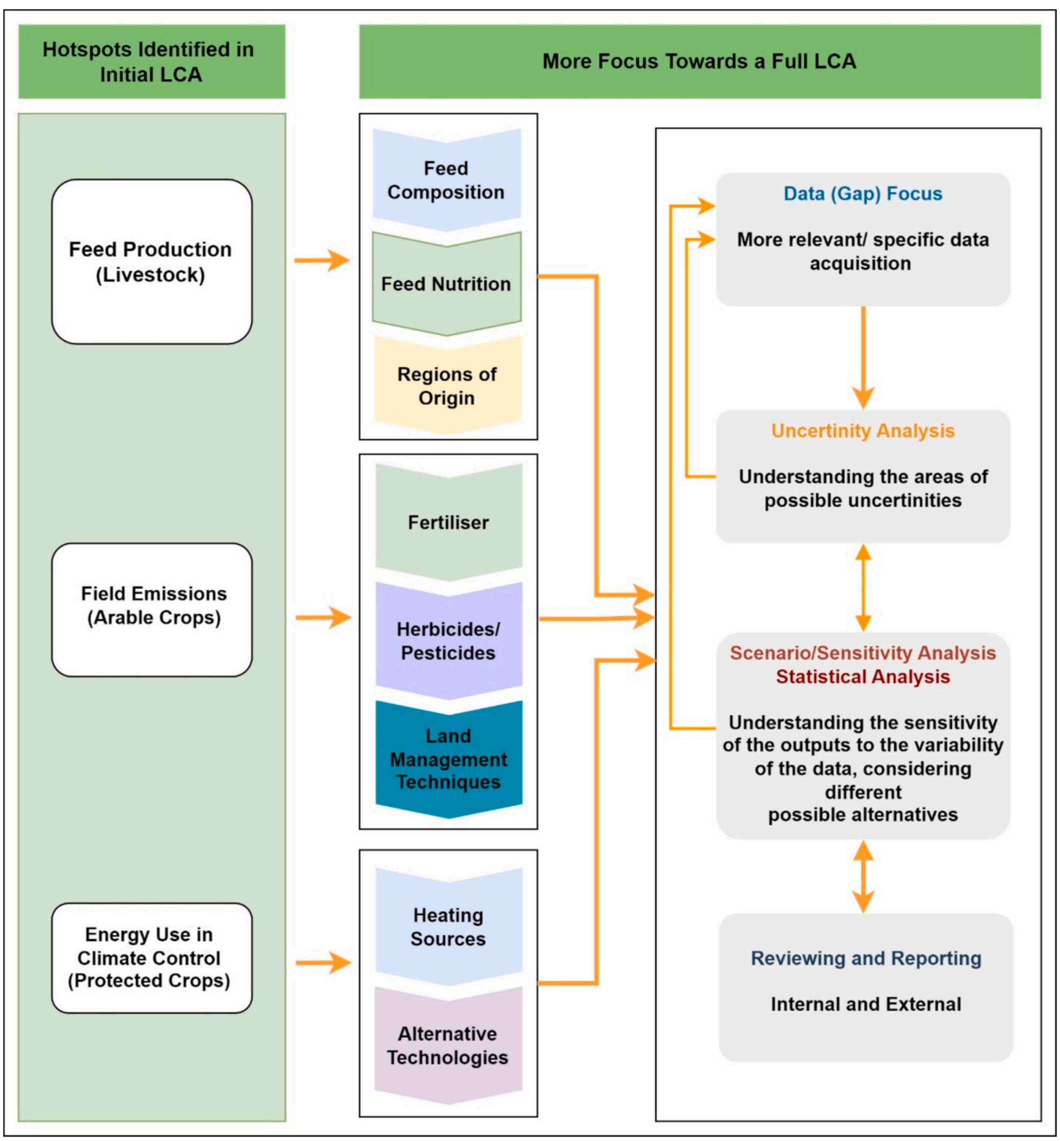


Fig. 7. Showcasing potential hotspots identified during the initial LCA and likely parameters for an in-depth study for a full LCA.

Table 11
A summary of potentially misleading approaches in an initial LCA with examples and strategies.

Life Cycle Stage	Misleading approaches	Examples and Consequences	How to correct/ prevent
Goal and Scope Definition	Defining imprecise FUs -Comparing multifunctional products and processes	-Comparing the environmental impacts of locally grown and imported products without considering the difference in production systems, transport, and storage, etc., would lead to biased conclusions.	Describe the FU properly/ transparently, including important quality aspects. Try finding comparable properties among alternative commodities.
	-Comparing products with different FUs.	e.g. Comparing UK heated greenhouse tomatoes and tomatoes grown in plastic tunnels in Spain. -Comparing absolute impacts without considering the actual functionality would lead to misleading interpretation and conclusions. e.g. Comparing freshly harvested fruit with fruit that has been briefly chilled on the farm.	Assess substitutability/ comparability of FU within the wider food system/value chain.
	Defining an imprecise system boundary -Fully or partially excluding upstream activities -Confusing starting and end points	Ignoring material production and, in some cases, residue or waste management, presents a partial picture of the overall process. e.g. Misrepresenting the specific type of plastic used in polytunnels (type, quantity, functional life) would give a wrong interpretation of GWP due to tomato production. Ignoring land use change in arable cropping.	Define the boundary explicitly. Communicate the rationale behind the specific boundary (was it due to data availability, resource constraints, or specific research questions, etc.) Compare results w/wo land use change,
LCI	-Use of outdated data -Use of data that represents a different geographical scope -Use of proxy with different products	-Over/under estimation of impacts -Use of banned pesticides/chemicals in some regions. e.g. including the use of Chlorpropham (which was banned in the UK in 2019) for potato storage. -Use secondary data available for certain countries as proxies for other regions. -Using generic data for fertiliser practices might not reflect variations on cultivar or regional agronomic practices.	Improve the temporal, geographical appropriateness of data For the same impact categories, compare hotspots with alternatives from other studies. Review input assumptions. Expand the choices of secondary data sources when primary datasets are not available. Utilise national statistics where possible as the first choice. Proper documenting data sources and assumptions. Seek stakeholder engagement where possible.
	-Double counting	-Use of aggregate estimates (e.g. fuel use) in conjunction with individual estimates for specific processes. -Misrepresentation of value chain stages, e.g. additional/ unnecessary storage processes. -Use of fertiliser emission factors that incorporate both production and field emissions whilst calculating direct emissions separately.	Rely on more than one source when using secondary data/ proxies. Develop a process/value chain flow diagram in tandem with FU boundary designation.
	-Over/under estimation of emission factors	Using generic emission factors for agricultural practices (fertiliser application/ manure management) without considering the specific climate conditions, soil types, cultivar, animal breeds, etc. e.g. Using water requirement/ consumption of UK field tomatoes as a proxy for Spain tomatoes would ignore the significant differences in irrigation needs. e.g. Using generic data for fertiliser usage without considering the cultivar, regional agronomic conditions.	Expand the choices of secondary data sources when primary datasets are not available Proper documenting of data sources and assumptions.
	Imprecise assumptions in allocation	Disproportionate allocation of environmental burden to co-products.	Don't exclude herd/flock dynamics from calculation.

(continued on next page)

Table 11 (*continued*)

Life Cycle Stage	Misleading approaches	Examples and Consequences	How to correct/ prevent
		e.g. Allocation milk vs meat, wool vs meat in livestock production based on differing biophysical or economic criteria.	If using economic allocation ensure consistent price applies (i.e. farm gate vs market price, inconsistent time boundary). Test price assumptions for co-products of low volume but high products. Consider function of main product.
Impact Assessment	-Use of impact categories, without representing the scope	Focusing on impacts, such as climate change (globally induced) while ignoring local impacts, such as water use or land use, etc., when considering the overall environmental burdens in certain areas. This could lead to overlooking some relevant impacts in some areas while favouring some other systems. e.g. lower climate change impact and higher water use impact in Spain tomatoes.	Compare impact categories with on the context and objective of the study, consider importance of global, local, resource, human impacts etc. Be aware about the limitations and communicate them transparently.
Interpretation	Confusing absolute vs relative impact comparison	Without proper understanding of the context and limitations of the data results can be misconstrued. e.g. Direct comparison of products based on absolute values (without transparently communicating the limitations)	Use relative impact if the focus is to quantify/ understand the significance of the impact compared to a benchmark or another system. Understand potential uncertainties and limitations of the assessment when using absolute values.
	Misinterpretation of uncertainty checks/ comparison with literature	Make conclusions without considering the exact boundaries, assumptions, and data choices in other studies. e.g. for arable crops, some studies may include storage at the farm gate, some may not. Comparing the results of two systems (i.e. with and without storage) could misinterpret the comparison.	Foster awareness about the limitations of the focused study and communicate them transparently.
	Misinterpretation of impacts	Misinterpretation of higher GWP, overlooking the actual impact	Distinguish/ categories direct and indirect emissions e.g. distinguishing energy and biogenic sources of GWP
		Misinterpretation of ecotoxicity and human toxicity e.g. Freshwater Ecotoxicity' and 'Human toxicity – cancer - non cancer effect' reflect the potential impact on ecosystem and human health, but should not be used to conclude that product X is safer or healthier than product Y.	The toxicity of a chemical must be assessed by considering concentration in the specific receiving compartment, not just via a quantity as reported in LCI.
	Neglecting trade-offs between categories	Focusing on a single impact category result fails to convey the wider impact of material substitution. e.g. A lower GWP in Spanish produced tomatoes when compared with UK greenhouse systems, overlooking a higher water scarcity impact	Be cognisant of the functions of each impact assessment category when interpreting hotspots. Consider the importance of global vs local vs resource use impacts to the research objective. Ensure the hotspots for each of these are compared.

outcomes based on established ranges and probabilities, sensitivity analysis is subjective and focuses on specific parameters (the selection of which is influenced by hotspot analyses), which may be varied arbitrarily (Groen and Heijungs, 2017). However, these are by no means mutually exclusive and indeed can be seen as part of the same toolkit. For example, Helmes et al. (2022), in comparing the impact of bio and fossil derived plastic products, initially applied a contribution analysis to distinguish the most consequential processes, this allowed a subsequent sensitivity analysis for important variables (e.g. recycled plastic content) and an uncertainly analysis via Monte Carlo simulation incorporating elements such as the uncertainty range for energy efficiency. Whilst both sensitivity and uncertainty analyses seek to answer similar but subtly different questions, in this case both confirm the potential benefits of changing key variables such as the inclusion fraction of recycled plastic. More generally sensitivity and/or scenario analysis is arguably more appropriate for a scoping study, but it should be seen as improving the baseline estimate before seeking to answer counterfactual questions.

A comparison against existing studies can help position the scoping results and thereby allow for an understanding of the uncertainties within the range of results for comparative systems. As presented in Table 10, in line with exemplar case studies, Parajuli et al. (2021) reported a similar range of GWP for potatoes grown in the US (González-García et al., 2023) and UK fields (Blonk et al., 2022). Holka and Bieńkowski (2020) reported similar range values for GWP impact for Spanish and Polish winter wheat, respectively. Similarly, for eggs, and pork (Rudolph et al., 2018; Nguyen et al., 2011), we share a similar range of results. However, reported higher GWP in other cases, for example in beef (Herron et al., 2021), could be partly due to economic allocation based on the specific economic values, and division between grass and non-grass feed. Equally, we observe different results than (Williams et al., 2006) based on different data sources and boundaries. Consideration of absolute values should be done under caution and only for data quality controls, as absolute values derived from initial studies might possibly underestimate or overestimate the actual environmental footprint.

Communicating uncertainty is crucial to ensure transparency and credibility, even in an initial LCA. Qualitative evaluations of uncertainty levels can help prioritise future efforts to address it. When data/results are uncertain and unavailable, expert judgement can be used to fill gaps and provide insights into likely outcomes. Guidance on the main agri-food value chain stakeholders (as in Table 2S, supporting materials-A) can be used to identify potential experts. Distinguishing uncertainties/errors in each single unit process and considering lifecycle stages according to the FU is important, as these uncertainties are highly co-related and overlooked easily (Norris, 2002).

3.4. Transitioning from an initial to full LCA

Depending on the study objectives, an LCA may conclude at the scoping stage or serve as a foundation for more in-depth or alternative analyses. For example, the case studies presented identified feed production, field emissions, and energy use as major hotspots for livestock, field crops, and protected crop production, respectively (Fig. 5a and b). Insights from initial LCAs can inform the transition to a full LCA and may include the following considerations (Fig. 7).

The appearance of common hotspots across multiple impact categories warrants further investigation. For instance, earlier results showed that animal feed dominates several impact categories, suggesting the need to re-evaluate assumptions regarding feed composition and region of origin, particularly given the significant impact of LUC.

While uncertainty in scoping results may limit the insights gained, identifying sources of uncertainty and data gaps is a critical step in adding value to a full LCA. This may involve more detailed data collection or dedicated uncertainty and sensitivity analyses. For PBFPs, the accuracy and variability of yield estimates can significantly influence impact scores. For ABFPs, the consideration of co-products—especially their economic value when economic allocation is used—is essential.

The emergence of anomalous or counter-intuitive results can highlight areas requiring further analysis to confirm environmental burdens. These anomalies may arise in comparisons within the same system (e.g., regional differences) or across the literature. A basic understanding of the systems under review is necessary to distinguish results that require confirmation or correction. For example, one commodity originally included in this study (carrots) was excluded from the group of case studies presented, as initial LCI and Impact assessment results suggested an unexpected nitrogen efficiency estimate (i.e. kg N/kg). Therefore, this was identified as worthy of further analysis to confirm or amend.

By way of summary, Table 11 presents potentially misleading approaches alongside suggestions for logical assumptions that can improve the quality of outputs in both initial and full LCAs within AFPSs. The results of a preparatory study can serve as components or building blocks in databases that support subsequent LCAs. Given that all stages—from goal and scope definition to interpretation—are interlinked, identifying errors or misleading assumptions at the level of individual unit processes is essential.

If the concerns outlined earlier are addressed, insights from scoping studies and hotspot analyses in AFPSs can become valuable contributions to the scientific literature and support informed decision-making across relevant sectors. However, it is also important to recognise that while there is strong demand for initial LCAs in AFPSs, broader adoption of simplified methods—and the associated risks of justifiable or misleading assumptions—will closely depend on the specific unit processes and flows within each food value chain.

This study has presented recommendations for minimum requirements in conducting initial LCAs for representative AFPSs up to the farm or field gate. Future work could extend these recommendations to more diversified agri-food value chain configurations, including farm-to-factory, farm-to-fork, farm-to-end-of-life, or farm-to-farm (i.e., cradle-to-cradle) systems.

4. Conclusions

An initial LCA can serve as a starting point for understanding the environmental impact associated with AFPSs as an input into a wider assessment of their sustainability and thus help to direct the allocation of resources for more comprehensive assessments. Results of the exemplar scoping analysis based on UK conditions demonstrate the impacts and the potential hotspots of the considered systems from an agronomic perspective. Therefore, such a study could help stakeholders make necessary decisions on impact improvement opportunities and prioritise impact reduction actions without digging into a full LCA while understanding directions for a full LCA. Comparisons across different impact categories and value chain stages emphasise the context dependency of interpretation. In the example of tomatoes, comparison with Spanish production systems for loose tomatoes shows that the (relative) GWP benefits of non-heated systems come with increased local water and eutrophication burdens. Expanding the boundary to consider additional processing stages reduces the GWP differential but has a limited impact on eutrophication.

Given the high demand for LCA in AFPSs, many publications base their results on scoping studies. Poor assumptions and data quality choices in such studies could lead to scientifically (and commercially) misleading conclusions, as well as the fact that a scoping analysis generally does not include an uncertainty and/or sensitivity check or external reviews. There is, therefore, a clear need to formulate practical guidance and recommendations for conducting an initial LCA, highlighting crucial data needs and potential strategies to overcome identified barriers. This paper has shared perspectives on conducting a transparent and reliable scoping study, identifying the potential data gaps, limitations, and the complex nature of the AFPSs, using representative case studies which can be used to further develop more insightful LCAs.

CRediT authorship contribution statement

Indika Thushari: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Asma Jebari:** Writing – review & editing. **Alison Carswell:** Writing – review & editing, Writing – original draft. **Thomas C. Ponsioen:** Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Conceptualization. **Stephen Morse:** Writing – review & editing, Writing – original draft. **Richard Murphy:** Writing – review & editing, Writing – original draft. **Conor Walsh:** Writing – review & editing, Writing – original draft, Funding acquisition.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Conor Walsh reports financial support was provided by United Kingdom Department for Environment Food and Rural Affairs. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

The authors gratefully acknowledge the funding provided by the Department of Environment Food and Rural Affairs, under ECM: 65226; Comparative Life Cycle Assessment of Commodity Production; Identifying Opportunities for Sustainable Productivity Growth Across the Agri-Food Chain.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.agsy.2026.104668.

Data availability

Data will be made available on request.

References

Adewale, C., Higgins, S., Granatstein, D., Stöckle, C.O., Carlson, B.R., Zaher, U.E., Carpenter-Boggs, L., 2016. Identifying hotspots in the carbon footprint of a small scale organic vegetable farm. Agric. Syst. 149, 112–121. https://doi.org/10.1016/J.AGSY.2016.09.004.

AHDB, 2021. Potatoes Market Information.

AHDB, 2022a. Harvest Report Report 5 – Week 7 Week Ending Week Ending 23 August 2022.

AHDB, 2022b. Feeding Growing and Finishing Pigs | AHDB [WWW Document]. Agriculture and Horticulture Development Board. URL. https://ahdb.org.uk/knowledge-library/feeding-growing-and-finishing-pigs (accessed 8.13.25).

Amicarelli, V., Lagioia, G., Bux, C., 2021. Global warming potential of food waste through the life cycle assessment: an analytical review. Environ. Impact Assess. Rev. 91, 106677. https://doi.org/10.1016/J.EIAR.2021.106677.

Antón, A., Torrellas, M., Montero, J.I., Ruijs, M., Vermeulen, P., Stanghellini, C., 2012. Environmental impact assessment of Dutch tomato crop production in a Venlo glasshouse. Acta Hortic. 927, 781–792. https://doi.org/10.17660/ACTAHORTIC.2012.927.97.

Ardente, F., Cellura, M., 2012. Economic allocation in life cycle assessment. J. Ind. Ecol. 16, 387–398. https://doi.org/10.1111/J.1530-9290.2011.00434.X.

Arzoumanidis, I., Raggi, A., Petti, L., 2014. Considerations when applying simplified LCA approaches in the wine sector. Sustainability (Switzerland) 6, 5018–5028. https://doi.org/10.3390/su6085018.

Arzoumanidis, I., Salomone, R., Petti, L., Mondello, G., Raggi, A., 2017. Is there a simplified LCA tool suitable for the Agri-food industry? An assessment of selected tools. J. Clean. Prod. 149, 406–425. https://doi.org/10.1016/J.JCLEPRO.2017.02.059.

Bamber, N., Turner, I., Arulnathan, V., Li, Y., Zargar Ershadi, S., Smart, A., Pelletier, N., 2020. Comparing sources and analysis of uncertainty in consequential and attributional life cycle assessment: review of current practice and recommendations. Int. J. Life Cycle Assess. 25, 168–180. https://doi.org/10.1007/S11367-019-01663-1/TABLES/6.

Barahmand, Z., Eikeland, M.S., 2022. Life cycle assessment under uncertainty: a scoping review. World 3, 692–717. https://doi.org/10.3390/WORLD3030039.

Barthel, M., James, K., Guinness, J., Barker, C., 2014. UNEP/SETAC Life Cycle Initiative-Flagship Project 3a (Phase 1) Hotspots Analysis: Mapping of Existing Methodologies, Tools and Guidance and Initial Recommendations for the Development of Global Guidance.

Barthel, M., Fava, J.A., Harnanan, C.A., Strothmann, P., Khan, S., Miller, S., 2015. Hotspots analysis: providing the focus for action. In: Sonnemann, G., Margni, M. (Eds.), Life Cycle Management. Springer Netherlands, Dordrecht, pp. 149–167. https://doi.org/10.1007/978-94-017-7221-1_12.

Barthel, M., Fava, J., James, K., Hardwick, A., Khan, S., 2017. Hotspots Analysis an Overarching Methodological Framework and Guidance for Product and Sector Level Application.

Beemsterboer, S., Baumann, H., Wallbaum, H., 2020. Ways to get work done: a review and systematisation of simplification practices in the LCA literature. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-020-01821-w.

Bengoa, X., Vincent Rossi, Q., Sebastien Humbert, Q., Thomas Nemecek, Q., Jens Lansche, A., Patrik Mouron, A., Eliane Riedener, A., 2015. World Food LCA Database Methodological Guidelines for the Life Cycle Inventory of Agricultural Products.

Bessou, C., Basset-Mens, C., Tran, T., Benoist, A., 2013. LCA applied to perennial cropping systems: a review focused on the farm stage. Int. J. Life Cycle Assess. 18, 340–361. https://doi.org/10.1007/s11367-012-0502-z.

Blonk, H., Kool, A., Luske, B., Advies, T.P.-B.M., 2010. Methodology for Assessing Carbon Footprints of Horticultural Products researchgate.net.

Blonk, H., Tyszler, M., Van Paassen, M., Braconi, N., Draijer, N., Van Rijn, J., 2022. Agri-footprint 6 Methodology Report Part 1: Methodology and Basic Principles About us Title Agri-footprint 6 Methodology Report.

Blonk, H., Tyszler, M., Van Paassen, M., Braconi, N., Draijer, N., Van Rijn, J., 2023. Agri-footprint 6 Methodology Report Part 2: Description of Data About us Title Agri-footprint 6 Methodology Report.

Budzinski, K., Constable, D., D'Aquila, D., Smith, P., Madabhushi, S.R., Whiting, A., Costelloe, T., Collins, M., 2022. Streamlined life cycle assessment of single use technologies in biopharmaceutical manufacture. New Biotechnol. 68, 28–36. https://doi.org/10.1016/J.NBT.2022.01.002.

Burhan, S., 2018. GaBi Databases 2018 (Service Pack 36) in openLCA.

Casey, J.W., Holden, N.M., 2025. Quantification of GHG Emissions from Sucker-beef Production in Ireland. https://doi.org/10.1016/j.agsy.2005.11.008.

Casson, A., Zambelli, M., Giovenzana, V., Tugnolo, A., Pampuri, A., Vignati, S., Beghi, R., Guidetti, R., 2023. Simplified environmental impact tools for agri-food system: a systematic review on trends and future prospective. Environ. Impact Assess. Rev. 102, 107175. https://doi.org/10.1016/J.EIAR.2023.107175.

Castellani, V., Sala, S., Benini, L., 2017. Hotspots analysis and critical interpretation of food life cycle assessment studies for selecting eco-innovation options and for policy support. J. Clean. Prod. 140, 556–568. https://doi.org/10.1016/j.jclepro.2016.05.078.

Cederberg, C., Henriksson, M., Berglund, M., 2013. An LCA researcher's wish list – data and emission models needed to improve LCA studies of animal production. Animal 7, 212–219. https://doi.org/10.1017/S1751731113000785.

Cerutti, A.K., Beccaro, G.L., Bruun, S., Bosco, S., Donno, D., Notarnicola, B., Bounous, G., 2014. Life cycle assessment application in the fruit sector: state of the art and recommendations for environmental declarations of fruit products. J. Clean. Prod. 73, 125–135. https://doi.org/10.1016/J.JCLEPRO.2013.09.017.

Chen, X., Wilfart, A., Puillet, L., Aubin, J., 2017. A new method of biophysical allocation in LCA of livestock co-products: modeling metabolic energy requirements of body-tissue growth. Int. J. Life Cycle Assess. 22, 883–895. https://doi.org/10.1007/s11367-016-1201-y.

Chen, X., Matthews, H.S., Griffin, W.M., 2021. Uncertainty caused by life cycle impact assessment methods: case studies in process-based LCI databases. Resour. Conserv. Recycl. 172, 105678. https://doi.org/10.1016/J.RESCONREC.2021.105678.

Colomb, V., Ait Amar, S., Mens, C.B., Gac, A., Gaillard, G., Koch, P., Mousset, J., Salou, T., Tailleur, A., van der Werf, H.M.G., 2015. AGRIBALYSE®, the French LCI Database for agricultural products: high quality data for producers and environmental labelling⋆. OCL 22.

Corrado, S., Castellani, V., Zampori, L., Sala, S., 2018. Systematic analysis of secondary life cycle inventories when modelling agricultural production: a case study for arable crops. J. Clean. Prod. 172, 3990–4000. https://doi.org/10.1016/J.JCLEPRO.2017.03.179.

Costa, M.P., Chadwick, D., Saget, S., Rees, R.M., Williams, M., Styles, D., 2020. Representing crop rotations in life cycle assessment: a review of legume LCA studies. Int. J. Life Cycle Assess. 25, 1942–1956. https://doi.org/10.1007/S11367-020-01812-X.

Curran, M., Hellweg, S., Beck, J., 2014. Is there any empirical support for biodiversity offset policy? Ecol. Appl. 24, 617–632. https://doi.org/10.1890/13-0243.1.

De Baan, L., Alkemade, R., Koellner, T., 2013. Land use impacts on biodiversity in LCA: a global approach. Int. J. Life Cycle Assess. 18, 1216–1230. https://doi.org/10.1007/S11367-012-0412-0.

Defra, 2022. Agri-climate Report 2022.

Defra, 2023. National Stattistics-Food Statistics in Your Pocket.

Detzel, A., Krüger, M., Busch, M., Blanco-Gutiérrez, I., Varela, C., Manners, R., Bez, J., Zannini, E., 2022. Life cycle assessment of animal-based foods and plant-based protein-rich alternatives: an environmental perspective. J. Sci. Food Agric. 102, 5098–5110. https://doi.org/10.1002/JSFA.11417;PAGEGROUP:STRING:PUBLICATION.

Dominguez Aldama, D., Grassauer, F., Zhu, Y., Ardestani-Jaafari, A., Pelletier, N., 2023. Allocation methods in life cycle assessments (LCAs) of agri-food co-products and food waste valorization systems: systematic review and recommendations. J. Clean. Prod. 421, 138488. https://doi.org/10.1016/J.JCLEPRO.2023.138488.

Dowson, M., Grogan, M., Birks, T., Harrison, D., Craig, S., 2012. Streamlined life cycle assessment of transparent silica aerogel made by supercritical drying. Appl. Energy 97, 396–404. https://doi.org/10.1016/J.APENERGY.2011.11.047.

Efeca, 2019. 2019 UK Roundtable on Sustainable Soya: Annual Progress Report, 2019.

European Commission, 2010. ILCD Handbook-General Guide for Life Cycle Assessment-Detailed Guidance.

Fan, J., Liu, C., Xie, J., Han, L., Zhang, C., Guo, D., Niu, J., Jin, H., McConkey, B.G., 2022. Life cycle assessment on agricultural production: a mini review on methodology, application, and challenges. Int. J. Environ. Res. Public Health. https://doi.org/10.3390/ijerph19169817.

Ferguson, A.E., Ramanathan, U., 2020. Key factors of carbon footprint in the UK food supply chains: a new perspective of life cycle assessment. Int. J. Oper. Prod. Manag. 40, 945–970. https://doi.org/10.1108/IJOPM-06-2019-0478.

Frankowska, A., Jeswani, H.K., Azapagic, A., 2019. Environmental sustainability issues in the food-energy-water nexus in the UK vegetables sector: Energy and water consumption. Energy Procedia 161, 150–156. https://doi.org/10.1016/J.EGYPRO.2019.02.074.

Gavrilova, O., Leip, A., Dong, H., MacDonald, J.D., Gomez Bravo, C.A., Amon, B., Barahona Rosales, R., Prado, A.D., de Lima, M.A., Oyhantcabal, W., 2019. Emission from Livestock and Manure Management (2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories No).

Gheewala, S.H., Jungbluth, N., Notarnicola, B., Ridoutt, B., van der Werf, H., 2020. No simple menu for sustainable food production and consumption. Int. J. Life Cycle Assess. 25, 1175–1182. https://doi.org/10.1007/S11367-020-01783-Z/METRICS.

Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2018. Addressing crop interactions within cropping systems in LCA. Int. J. Life Cycle Assess. 23, 1735–1743. https://doi.org/10.1007/S11367-017-1393-9.

Goglio, P., Knudsen, M.T., Van Mierlo, K., Röhrig, N., Fossey, M., Maresca, A., Hashemi, F., Waqas, M.A., Yngvesson, J., Nassy, G., Broekema, R., Moakes, S., Pfeifer, C., Borek, R., Yanez-Ruiz, D., Cascante, M.Q., Syp, A., Zylowsky, T., Romero-Huelva, M., Smith, L.G., 2023. Defining common criteria for harmonizing life cycle assessments of livestock systems. Clean. Prod. Lett. 4, 100035. https://doi.org/10.1016/J.CLPL.2023.100035.

Goglio, P., Moakes, S., Knudsen, M.T., Van Mierlo, K., Adams, N., Maxime, F., Maresca, A., Romero-Huelva, M., Waqas, M.A., Smith, L.G., Grossi, G., Smith, W., De Camillis, C., Nemecek, T., Tei, F., Oudshoorn, F.W., 2024. Harmonizing methods to account for soil nitrous oxide emissions in Life Cycle assessment of agricultural systems. Agric. Syst. 219, 104015. https://doi.org/10.1016/J.AGSY.2024.104015.

González-García, S., Almeida, F., Brandão, M., 2023. Do carbon footprint estimates depend on the LCA modelling approach adopted? A case study of bread wheat grown in a crop-rotation system. Sustainability 15, 4941.

Gradin, K.T., Björklund, A., 2021. The common understanding of simplification approaches in published LCA studies—a review and mapping. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-020-01843-4.

Groen, E.A., Heijungs, R., 2017. Ignoring correlation in uncertainty and sensitivity analysis in life cycle assessment: what is the risk? Environ. Impact Assess. Rev. 62, 98–109.

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: past, present, and future. Environ. Sci. Technol. 45, 90–96. https://doi.org/10.1021/ES101316V/ASSET/IMAGES/LARGE/ES-2010-01316V_0003.JPEG.

Guo, M., Murphy, R.J., 2012. LCA data quality: sensitivity and uncertainty analysis. Sci. Total Environ. 435–436, 230–243. https://doi.org/10.1016/j.scitotenv.2012.07.006.

Hall, H.N., O'Neill, H.V.M., Scholey, D., Burton, E., Dickinson, M., Fitches, E.C., 2018. Amino acid digestibility of larval meal (Musca domestica) for broiler chickens. Poult. Sci. 97, 1290–1297.

Halpern, B.S., Cottrell, R.S., Blanchard, J.L., Bouwman, L., Froehlich, H.E., Gephart, J.A., Sand Jacobsen, N., Kuempel, C.D., McIntyre, P.B., Metian, M., Moran, D.D., 2019. Putting all foods on the same table: achieving sustainable food systems requires full accounting. Proc. Natl. Acad. Sci. 116 (37), 18152–18156.

Heidari, M.D., Mathis, D., Blanchet, P., Amor, B., 2019. Streamlined life cycle assessment of an innovative bio-based material in construction: a case study of a phase change material panel. Forests 10, 160. https://doi.org/10.3390/F10020160.

Heijungs, R., 2021. Selecting the best product alternative in a sea of uncertainty. Int. J. Life Cycle Assess. 26, 616–632. https://doi.org/10.1007/s11367-020-01851-4.

Helmes, R.J., Huijbregts, M.A., Henderson, A.D., Jolliet, O., 2012. Spatially explicit fate factors of phosphorous emissions to freshwater at the global scale. Int. J. Life Cycle Assess. 17 (5), 646–654.

Helmes, R.J., Goglio, P., Salomoni, S., van Es, D.S., Vural Gursel, I., Aramyan, L., 2022. Environmental impacts of end-of-life options of biobased and fossil-based polyethylene terephthalate and high-density polyethylene packaging. Sustainability 14 (18), 11550.

Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., del Prado, A., Kasimir, Å., MacDonald, J.D., Ogle, S.M., Regina, K., van der Weerden, T.J., 2019. N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application.

Herron, J., Curran, T.P., Moloney, A.P., McGee, M., O'Riordan, E.G., O'Brien, D., 2021. Life cycle assessment of pasture-based suckler steer weanling-to-beef production systems: effect of breed and slaughter age. Animal 15, 100247.

Hoekstra, A.Y., Mekonnen, M.M., 2012. The water footprint of humanity. Proc. Natl. Acad. Sci. USA 109, 3232–3237. https://doi.org/10.1073/PNAS.1109936109.

Holka, M., Bieńkowski, J., 2020. Assessment of environmental burdens of winter wheat production in different agrotechnical systems. Agronomy 10, 1303.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J. Life Cycle Assess. 22, 138–147. https://doi.org/10.1007/S11367-016-1246-Y/TABLES/2.

Hung, C.R., Ellingsen, L.A.W., Majeau-Bettez, G., 2020. LiSET: a framework for early-stage life cycle screening of emerging technologies. J. Ind. Ecol. 24, 26–37. https://doi.org/10.1111/jiec.12807.

Igos, E., Benetto, E., Meyer, R., Baustert, P., Othoniel, B., 2019. How to treat uncertainties in life cycle assessment studies? Int. J. Life Cycle Assess. 24, 794–807.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Calvo Buendia, E., Tanabe, K., Kranjc, A., Baasansuren, J., Fukuda, M., Ngarize, S., Osako, A., Pyrozhenko, Y., Shermanau, P. and Federici, S. (eds). IPCC, Switzerland [WWW Document]. https://www.ipcc.ch/report/2019-refinement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/. URL https://www.ipcc.ch/2019/ (accessed 8.12.25).

IPCC, 2021. Climate change 2021-the physical science basis. In: Working Group 1 of the Sixth Assessment Report of the Intergovernmental Panel on Climate Change.

ISO, 2006a. ISO 14040: Environmental Management Life Cycle Assessment Principles and Framework. Switzerland.

ISO, 2006b. ISO 14044: Environmental Management Life Cycle Assessment Requirements and Management. Switzerland.

Joshi, M., Srivastava, R.K., Vahlne, N., Ahlgren, E.O., Bruce, N., Perez-Padilla, R., Albalak, R., IPCC, US EPA Center for Corporate Climate Leadership, O. A R, Herold, A., Saxena, N.C., Habermehl, H., Fao Jeuland, M.A., Pattanayak, S.K., Fuels, C., Garg, A., Mohan, P., Edwards, R., Karnani, S., Fisher, E.M., Johnson, M., Naeher, L., Smith, K.R., Morawska, L., Gómez, D.R., Watterson, J.D., Americanohia, B.B., Ha, C., Marland, G., Matsika, E., Namayanga, L.N., Osman-Elasha, B., Saka, J.D.K., Treanton, K., National Council for Air and Stream Improvement Inc. (NCASI), Kavi Kumar, K.S., Viswanathan, B., Mt, G.J., Helena, S., 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Genebra, Suíça, 66, p. 44. https://doi.org/10.1209/epl/i2005-10515-2.

Jungbluth, N., Frischknecht, R., 2010. Cumulative energy demand. In: Implementation of Life Cycle Impact Assessment Methods, pp. 33–40.

Kebreab, E., Liedke, A., Caro, D., Deimling, S., Binder, M., Finkbeiner, M., 2016. Environmental impact of using specialty feed ingredients in swine and poultry production: a life cycle assessment1. J. Anim. Sci. 94, 2664–2681. https://doi.org/10.2527/jas.2015-9036.

Kiehbadroudinezhad, M., Hosseinzadeh-Bandbafha, H., Tajuddin, Sheikh Ahmad, Tabatabaei, M., Aghbashlo, M., 2025. A critical review of life cycle assessment of renewable agricultural systems. Sustain. Energy Technol. Assess. 73, 104100. https://doi.org/10.1016/J.SETA.2024.104100.

Kiemel, S., Rietdorf, C., Schutzbach, M., Miehe, R., 2022. How to simplify life cycle assessment for industrial applications—a comprehensive review. Sustainability 14, 15704. https://doi.org/10.3390/SU142315704.

Klöpffer, W., Grahl, B., 2014. Life Cycle Assessment (LCA): A Guide to Best Practice.

Koch, P., 2015. Agribalyse®: Methodology. ADEME, Angers, France.

Kounina, A., Margni, M., Bayart, J.B., Boulay, A.M., Berger, M., Bulle, C., Frischknecht, R., Koehler, A., Canals, Milà I., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., Van Zelm, R., Verones, F., Humbert, S., 2013. Review of methods addressing freshwater use in life cycle inventory and impact assessment. Int. J. Life Cycle Assess. 18, 707–721. https://doi.org/10.1007/s11367-012-0519-3.

Laurent, A., Weidema, B.P., Bare, J., Liao, X., Maia de Souza, D., Pizzol, M., Sala, S., Schreiber, H., Thonemann, N., Verones, F., 2020. Methodological review and detailed guidance for the life cycle interpretation phase. J. Ind. Ecol. 24, 986–1003. https://doi.org/10.1111/JIEC.13012;PAGEGROUP:STRING:PUBLICATION.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012a. Predicting the environmental impacts of chicken systems in the United Kingdom through a life cycle assessment: egg production systems. Poult. Sci. 91, 26–40.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012b. Predicting the environmental impacts of chicken systems in the United Kingdom through a life cycle assessment: broiler production systems. Poult. Sci. 91, 8–25.

Leinonen, I., Williams, A.G., Kyriazakis, I., 2014. The effects of welfare-enhancing system changes on the environmental impacts of broiler and egg production. Poult. Sci. 93, 256–266.

Liu, M., Zhu, G., Tian, Y., 2024. The historical evolution and research trends of life cycle assessment. Green Carbon 2, 425–437. https://doi.org/10.1016/J.GREENCA.2024.08.003.

Mackenzie, S.G., Leinonen, I., Kyriazakis, I., 2017. The need for co-product allocation in the life cycle assessment of agricultural systems—is "biophysical" allocation progress? Int. J. Life Cycle Assess. 22, 128–137. https://doi.org/10.1007/s11367-016-1161-2.

MacLeod, M.J., Vellinga, T., Opio, C., Falcucci, A., Tempio, G., Henderson, B., Makkar, H., Mottet, A., Robinson, T., Steinfeld, H., Gerber, P.J., 2018. Invited review: a position on the Global Livestock Environmental Assessment Model (GLEAM). Animal 12, 383–397. https://doi.org/10.1017/S1751731117001847.

Majewski, E., Komerska, A., Kwiatkowski, J., Malak-Rawlikowska, A., Was, A., Sulewski, P., Gołą, M., Pogodzinska, K., Lecoeur, J.L., Tocco, B., Török, Á., Donati, M., Vittersø, G., 2020. Are short food supply chains more environmentally sustainable than long chains? A life cycle assessment (LCA) of the eco-efficiency of food chains in selected EU countries. Energies 13, 4853. https://doi.org/10.3390/EN13184853.

McAuliffe, G.A., Takahashi, T., Mogensen, L., Hermansen, J.E., Sage, C.L., Chapman, D. V., Lee, M.R.F., 2017. Environmental trade-offs of pig production systems under varied operational efficiencies. J. Clean. Prod. 165, 1163–1173. https://doi.org/10.1016/J.JCLEPRO.2017.07.191.

McAuliffe, G.A., Takahashi, T., Lee, M.R.F., 2018. Framework for life cycle assessment of livestock production systems to account for the nutritional quality of final products. Food Energy Secur. 7, e00143. https://doi.org/10.1002/FES3.143;SUBPAGE:STRING:FRAMEDPDF;PAGE:STRING:ARTICLE/CHAPTER.

McAuliffe, G.A., Takahashi, T., Lee, M.R.F., 2020. Applications of nutritional functional units in commodity-level life cycle assessment (LCA) of agri-food systems. Int. J. Life Cycle Assess. 25, 208–221. https://doi.org/10.1007/S11367-019-01679-7/TABLES/1.

McAuliffe, Graham A., Lynch, J., Cain, M., Buckingham, S., Rees, R.M., Collins, A.L., Allen, M., Pierrehumbert, R., Lee, M.R.F., Takahashi, T., 2023a. Are single global warming potential impact assessments adequate for carbon footprints of agri-food systems? Environ. Res. Lett. 18, 084014.

McAuliffe, G.A., Takahashi, T., Beal, T., Huppertz, T., Leroy, F., Buttriss, J., Collins, A.L., Drewnowski, A., McLaren, S.J., Ortenzi, F., van der Pols, J.C., van Vliet, S., Lee, M.R. F., 2023b. Protein quality as a complementary functional unit in life cycle assessment (LCA). Int. J. Life Cycle Assess. 28, 146–155. https://doi.org/10.1007/S11367-022-02123-Z/FIGURES/1.

McAuliffe, G.A., Beal, T., Lee, M.R.F., van der Pols, J.C., 2024. Editorial: pushing the frontiers of nutritional life cycle assessment (nLCA) to identify globally equitable and sustainable agri-food systems. Front. Sustain. Food Syst. https://doi.org/10.3389/fsufs.2024.1471102.

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agric. Syst. 104, 217–232. https://doi.org/10.1016/J.AGSY.2010.10.002.

Nemecek, T., Hayer, F., Bonnin, E., Carrouée, B., Schneider, A., Vivier, C., 2015. Designing eco-efficient crop rotations using life cycle assessment of crop combinations. Eur. J. Agron. 65, 40–51. https://doi.org/10.1016/J.EJA.2015.01.005.

Nemecek, T., Bengoa, X., Lansche, J., Mouron, P., Riedener, E., Humbert, S., 2019. Methodological guidelines for the life cycle inventory of agricultural products. In: Version 3.5, December 2019. World Food LCA Database (WFLDB). Quantis and Agroscope, Lausanne and Zurich, Switzerland.

Nguyen, T.L.T., Hermansen, J.E., Mogensen, L., 2011. Environmental Assessment of Danish Pork. Aarhus University, Aarhus, Denmark.

Nicoletti, G., Notarnicola, B., 1999. Streamlining LCA as a tool to evaluate the sustainability of commodities production processes. Annali di Chimica (Rome) 89.

Norris, G., 2002. The Many Dimensions of Uncertainty Analysis in LCA.

Notarnicola, B., Hayashi, K., Curran, M.A., Huisingh, D., 2012. Progress in working towards a more sustainable agri-food industry. J. Clean. Prod. 28, 1–8. https://doi.org/10.1016/J.JCLEPRO.2012.02.007.

Notarnicola, B., Tassielli, G., Renzulli, P.A., Giudice, A. Lo, 2015. Life Cycle Assessment in the agri-food sector: an overview of its key aspects, international initiatives, certification, labelling schemesand methodological issues. In: Life Cycle Assessment in the Agri-food Sector: Case Studies, Methodological Issues and Best Practices, pp. 1–56. https://doi.org/10.1007/978-3-319-11940-3_1.

Notarnicola, B., Sala, S., Anton, A., McLaren, S.J., Saouter, E., Sonesson, U., 2017. The role of life cycle assessment in supporting sustainable agri-food systems: a review of the challenges. J. Clean. Prod. 140. https://doi.org/10.1016/j.jclepro.2016.06.071.

Notarnicola, B., Tassielli, G., Renzulli, P.A., Di Capua, R., Saija, G., Salomone, R., Primerano, P., Petti, L., Raggi, A., Casolani, N., Strano, A., Mistretta, M., 2024. Life cycle inventory data for the Italian agri-food sector: background, sources and methodological aspects. Int. J. Life Cycle Assess. 29, 1430–1445. https://doi.org/10.1007/S11367-021-02020-X/TABLES/2.

Panesar, D.K., Seto, K.E., Churchill, C.J., 2017. Impact of the selection of functional unit on the life cycle assessment of green concrete. Int. J. Life Cycle Assess. 22, 1969–1986. https://doi.org/10.1007/S11367-017-1284-0/TABLES/8.

Parajuli, R., Matlock, M.D., Thoma, G., 2021. Cradle to grave environmental impact evaluation of the consumption of potato and tomato products. Sci. Total Environ. 758, 143662.

Pelletier, N., Ardente, F., Brandão, M., De Camillis, C., Pennington, D., 2015. Rationales for and limitations of preferred solutions for multi-functionality problems in LCA: is increased consistency possible? Int. J. Life Cycle Assess. 20, 74–86. https://doi.org/10.1007/S11367-014-0812-4/TABLES/1.

Pérez, R., Argüelles, F., Laca, Amanda, Laca, Adriana, 2024. Evidencing the importance of the functional unit in comparative life cycle assessment of organic berry crops. Environ. Sci. Pollut. Res. 31, 22055–22072. https://doi.org/10.1007/S11356-024-32540-6.

Pernollet, F., Coelho, C.R.V., van der Werf, H.M.G., 2017. Methods to simplify diet and food life cycle inventories: accuracy versus data-collection resources. J. Clean. Prod. 140, 410–420. https://doi.org/10.1016/J.JCLEPRO.2016.06.111.

Ponsioen, T.C., van der Werf, H.M.G., 2017. Five propositions to harmonize environmental footprints of food and beverages. J. Clean. Prod. 153, 457–464. https://doi.org/10.1016/j.jclepro.2017.01.131.

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science 1979 (360), 987–992. https://doi.org/10.1126/science.aaq0216.

Porto Costa, M., Saget, S., Zimmermann, B., Petig, E., Angenendt, E., Rees, R.M., Chadwick, D., Gibbons, J., Shrestha, S., Williams, M., Styles, D., 2023. Environmental and land use consequences of replacing milk and beef with plant-based alternatives. J. Clean. Prod. 424, 138826. https://doi.org/10.1016/J.JCLEPRO.2023.138826.

PotatoPro, 2023. Production and Consumption.

Roy, P.-O., Azevedo, L.B., Margni, M., van Zelm, R., Deschênes, L., Huijbregts, M.A.J., 2014. Characterization factors for terrestrial acidification at the global scale: a systematic analysis of spatial variability and uncertainty. Sci. Total Environ. 500, 270–276.

RSPCA, 2020. The life of a typical meat chicken. Royal Society for the Prevention of Cruelty to Animals.

Rudolph, G., Hörtenhuber, S., Bochicchio, D., Butler, G., Brandhofer, R., Dippel, S., Dourmad, J.Y., Edwards, S., Früh, B., Meier, M., 2018. Effect of three husbandry systems on environmental impact of organic pigs. Sustainability 10, 3796.

Sala, S., Laurent, A., Vieira, M., Van Hoof, G., 2020. Implications of LCA and LCIA choices on interpretation of results and on decision support. Int. J. Life Cycle Assess. 25, 2311–2314. https://doi.org/10.1007/S11367-020-01845-2/TABLES/1.

Siol, C., Majer, S., Thrän, D., 2025. Integrating soil- and agro-ecosystem models into life cycle assessments of sustainable management of agricultural residues: a review in the context of sustainable development goals and planetary boundaries. Int. J. Life Cycle Assess. 2025, 1–17. https://doi.org/10.1007/S11367-025-02550-8.

Soussana, J.F., 2014. Research priorities for sustainable agri-food systems and life cycle assessment. J. Clean. Prod. 73, 19–23. https://doi.org/10.1016/J.JCLEPRO.2014.02.061.

Soust-Verdaguer, B., Llatas, C., García-Martínez, A., 2016. Simplification in life cycle assessment of single-family houses: a review of recent developments. Build. Environ. 103, 215–227. https://doi.org/10.1016/J.BUILDENV.2016.04.014.

Spindler, C., Ciroth, A., 2022. Ecoinvent 3.9.1 in openLCA.

Stocker, T.F., Qin, D., Plattner, G.K., Tignor, M.M.B., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., 2014. Climate change 2013 – the physical science basis: working Group I contribution to the fifth assessment report of the intergovernmental panel on climate change. In: Climate Change 2013 the Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change 9781107057999, pp. 1–1535. https://doi.org/10.1017/CBO9781107415324.

Timpanaro, G., Branca, F., Cammarata, M., Falcone, G., Scuderi, A., 2021. Life cycle assessment to highlight the environmental burdens of early potato production. Agronomy 11, 879.

USDA, 2023. Knowledge Article: What is the Most Consumed Meat in the World? [WWW Document]. U.S. Department of Agriculture.

van Paassen, M., Braconi, N., Kuling, L., Durlinger, B., Gual, P., 2019. Agri-footprint 5.0. Blonk Consultants: Gouda, The Netherlands, p. 134.

van Zelm, R., Stam, G., Huijbregts, M.A.J., van de Meent, D., 2013. Making fate and exposure models for freshwater ecotoxicity in life cycle assessment suitable for organic acids and bases. Chemosphere 90, 312–317. https://doi.org/10.1016/J.CHEMOSPHERE.2012.07.014.

van Zelm, R., Preiss, P., van Goethem, T., Van Dingenen, R., Huijbregts, M., 2016. Regionalized life cycle impact assessment of air pollution on the global scale: damage to human health and vegetation. Atmos. Environ. 134, 129–137. https://doi.org/10.1016/J.ATMOSENV.2016.03.044.

Vieira, M.D.M., Ponsioen, T.C., Goedkoop, M.J., Huijbregts, M.A.J., 2017. Surplus ore potential as a scarcity indicator for resource extraction. J. Ind. Ecol. 21, 381–390. https://doi.org/10.1111/JIEC.12444.

Wang, Y., Levis, J.W., Barlaz, M.A., 2021. Development of streamlined life-cycle assessment for the solid waste management system. Environ. Sci. Technol. 55, 5475–5484. https://doi.org/10.1021/ACS.EST.0C07461/SUPPL_FILE/ES0C07461_SI_001.PDF.

Webb, J., Williams, A.G., Hope, E., Evans, D., Moorhouse, E., 2013. Do foods imported into the UK have a greater environmental impact than the same foods produced within the UK? Int. J. Life Cycle Assess. 18, 1325–1343. https://doi.org/10.1007/s11367-013-0576-2.

Weidema, B.P., 2013. Overview and methodology. Data quality guideline for the ecoinvent database version 3. Ecoinvent Report 1 (V3). The ecoinvent Centre, St. Gallen.

Weidema, B., 2014. Has ISO 14040/44 failed its role as a standard for life cycle assessment? J. Ind. Ecol. 18, 324. https://doi.org/10.1111/JIEC.12139.

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/S11367-016-1087-8/FIGURES/7.

WFLDB, 2022. Quantis - WFLDB - World Food Life Cycle Assessment Database.

Wilfart, A., Gac, A., Salaün, Y., Aubin, J., Espagnol, S., 2021. Allocation in the LCA of meat products: is agreement possible? Clean. Environ. Syst. 2, 100028. https://doi.org/10.1016/J.CESYS.2021.100028.

Williams, A.G., Audsley, E., Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main Report. Defra Research Project IS0205.

Williams, A.G., Audsley, E., Sandars, D.L., 2010. Environmental burdens of producing bread wheat, oilseed rape and potatoes in England and Wales using simulation and system modelling. Int. J. Life Cycle Assess. 15, 855–868. https://doi.org/10.1007/S11367-010-0212-3/TABLES/15.

WMO, 2010. Scientific assessment of ozone depletion: 2010. In: Global Ozone Research and Monitoring Project-report No.52.

Zampori, L., Pant, R., 2019. Suggestions for Updating the Organisation Environmental Footprint (OEF) Method. Publications Office of the European Union.

Zampori, L., Saouter, E., Schau, E., Cristobal, J., Castellani, V., Sala, S., 2016. Guide for Interpreting Life Cycle Assessment Result. Publications Office of the European Union, Luxembourg.