Report containing the harmonization of LCA methodologies for livestock systems

April 2025

Pathways FOR SUSTAINABLE FOOD

This project has received funding from the European Union's Horizon 2020 Research and Innovation Programme under grant agreement No 101000395.

Deliverable X.X	Title
WP	5
Deliverable lead	UniPG
Authors	Pietro Goglio, Simon Moakes, Simone Pelaracci
Contact	pietro.goglio@unipg.it
Reviewer	David Yanez Ruiz, Thomas Nemecek
GA number	101000395
Funding body	H2020
Start / duration	1/9/2021 / 60 months
Type of deliverable (R, DEM, DEC, OTHER, ETHICS, ORDP, DATA)[1]	R
Dissemination level (PU, PP, RE, CO)[2]	PU
Website	https://pathways-project.com/

Version	Date	Description	Author
1	23/01/2025	Finalised version	Pietro Goglio

[1] Document, report (R), Demonstrator, pilot, prototype (DEM), Websites, patent filings, videos, etc., OTHER, Ethics requirement (ETHICS), Open Research Data Pilot (ORDP), Data sets, microdata, etc. (DATA)
[2] Public (PU), Restricted to other programme participants including EC (PP), Restricted to a group specified by the consortium including the EC (RE), Confidential, only for members of the consortium including EC (CO)

Table of Contents

List of Figures

List of Tables

Executive Summary

This deliverable contains the methods and outcomes from a harmonization and review of LCA methodological aspects related to the assessment of livestock production. Due to the potentially wide focus of LCA in livestock systems, an anonymous survey of LCA experts was carried out. Within the survey, each participant was asked to provide a preference value to enable ranking of topics and provide a priority list. The results of the survey were then used as the selection basis for five topics including, circularity (SG1), biodiversity (SG2), animal welfare (SG3), nutrition (SG4) and greenhouse gases (SG5). The SG5 topic was further divided into 4 sub-topic areas including soil carbon (SG5a), nitrous oxide (SG5b), manure handling (SG5c and enteric emissions (SG5d).
Utilising a modified DELPHI method, authors within each topic developed general and specific criteria through a participatory approach that includes several workshops among 21 LCA experts and two anonymous surveys. The general criteria were assigned to assess the quality of the reviewed method as an LCA methodology. Whilst the specific criteria were developed to evaluate the ability of the method to provide a comprehensive assessment and were unique to each topic. The developing processes and results of both types of criteria are described in detail in Goglio et al. (2023). For each of the key topics described in the deliverable a set of LCA recommendations are provided in Table 1.

Within SG1 (**circularity**), the agricultural sector faces increasing challenges to reduce environmental impacts while meeting global food demands. These necessary changes are an opportunity to redefine functions of the livestock sector by moving its traditionally linear structure towards Circular Pathways (CP). Currently, common assumptions related to Life Cycle Assessment (LCA), predominantly method used to assess potential environmental impacts, only partially addresses CP. SG1 section provides a critical assessment of current LCA based methods used to evaluate the environmental impacts of the livestock sector. In this harmonization and literature review, it was found that there is a need to improve and harmonize LCA methodologies to have greater coherence for LCA applied to the livestock sector. Encompassing circularity concepts such as (i) competition of use of products (food, feed, fuel and biomaterial use), (ii) closing nutrients cycles (crop-livestock interaction) and (iii) economic and social considerations (multi-functionality) in a single assessment remains a challenge. The analysis showed multi-functionality and associated issues such as co-products, by-products and waste considerations as keystones in circularity in LCA. Areas of developments needed to reach better methodological compliance between level of accuracy and applicability were identified and several LCA approaches, such as combined approach covered themes not considered in standard LCA, seem relevant to capture CP dimensions. Combined approaches need to be further developed to reduce variability.

The SG2 (**biodiversity**) topic review found that whilst agricultural intensification and expansion have significantly contributed to global biodiversity loss, primarily through land management changes. This is also affected by livestock systems. However, there is no consensus on how LCA should assess these biodiversity impacts. This systematic review evaluates existing LCA methods for biodiversity impact assessment, comparing expert scoring-based (ESB) and biodiversity indicator-based (BIB) methods to identify research gaps and methodological improvements.

Results indicate that BIB methods generally outperform ESB methods in robustness and completeness, as they rely on biodiversity models rather than expert opinions and evidence based scoring. However, BIB methods struggle to capture specific land management practices, whereas ESB methods offer more flexibility in evaluating these impacts. The available methods focus on various biodiversity levels and aspects, but each considers limited biodiversity characteristics and cannot represent the comprehensive biodiversity concept. BIB methods tend to use land management intensity levels, while ESB methods focus on specific land management practices. Despite their advantages, neither approach is sufficient for fully capturing biodiversity impacts across supply chains For future studies, it is advisable to (1) model the direct (on-farm) impacts of land management change at the midpoint level; (2) establish cause-effect relationships between crucial land management practices and biodiversity indicators and distinguish between direct (on-site) and indirect (off-site) biodiversity impacts resulting from land management change; (3) characterise land-use intensity levels based on specific land management practices and include the positive impact from agroecological practices. This review highlights the current state of LCA methods and suggests improvements to better account for the complexity of biodiversity impacts from agricultural land management.

The SG3 (**animal welfare**) topic review highlighted Life Cycle Assessment (LCA) is a valuable tool for evaluating the environmental impacts of livestock systems, but the integration of animal welfare remains limited. This review focused on studies that integrated animal welfare and life cycle assessment (LCA), selecting only peer-reviewed research related to livestock farming published in English after 2012. Eleven methods were evaluated based on a set of established general LCA criteria: credibility, transparency and reproducibility, fairness and acceptance, robustness, and applicability. In addition, specific criteria for incorporating animal welfare into LCA were applied, including accuracy, which reflects the ability to assess welfare across diverse production systems, and coherence, which refers to relevance across all stages of an animal's life.

The study found very few methods that integrate animal welfare assessments with LCA, with methodological complexity and data collection forming key barriers. Most standard LCAs integrating animal welfare focussed on few and easily attainable indicators with a limited connection to the functional unit, which limited their accuracy and prevented adequate coverage of the complexity of animal welfare. Social LCAs tended to perform better due to increased numbers of indicators covering wider animal welfare topics. Utilising approaches from social LCAs while ensuring the functional unit is linked to all indicators could allow standard LCA to accurately integrate animal welfare.

The SG4 (**nutritional**) topic study aimed to identify the most appropriate method(s) to integrate nutritional parameters in LCA. The LCA method development to compare the environmental impacts of dietary changes was excluded from this research which focused only on how nutritional aspects were included in the LCA of livestock product. Literature indicates that it is important to include the nutritional aspects, but there is little consensus on the preferred method.
A systematic literature review and screening of relevant studies was used to identify related methods that integrate nutritional aspects with functional units (FU). The identified studies were scored, based on the general criteria defined by means of a literature review of LCA frameworks and expert workshops. These methods were then subsequently scored based on the specific criteria including, "coverage of multiple nutrients", "consideration of human nutritional requirements" and "accuracy". Based on this approach, 16 high scoring methods were selected. Of these, 4 methods were deemed the most appropriate, based on their inclusion of multiple nutrients within a FU and were able to objectively calculate the nutrient scores. Furthermore, recommendations were formulated to test these methods in different contexts to be able to identify the most appropriate method.

The SG5a (**soil carbon**) methodological review of LCA of livestock systems encompassing crop-livestock interaction. It was discussed that effective mitigation strategies of livestock production should consider a better crop-livestock interaction. Soil organic carbon (SOC) sequestration plays a critical role in reducing atmospheric GHG concentrations, yet it is often underrepresented in Life Cycle Assessment (LCA) methodologies for agricultural systems due to challenges in accurately accounting for soil carbon dynamics. This study aimed to evaluate and harmonize soil carbon estimation tools, including simulation models, emission factors, and direct measurements, to better integrate SOC sequestration into LCAs.
A systematic review identified 263 relevant studies from an initial pool of 29,151, ultimately analyzing 20 tools categorized by complexity and data requirements. Using expert workshops and a participatory approach, each tool was evaluated against established criteria. Results revealed a trade-off between applicability and accuracy.This emphasizes the importance of selecting tools based on LCA objectives, available data, and practitioner expertise.
Key challenges include the influence of initial SOC levels, time perspectives for assessments, and the complexities of soil dynamics under varying agricultural practices. The findings underscore the need for improved LCA methodologies that can balance accuracy and applicability while addressing data

and expertise constraints. Recommendations highlight the importance of aligning tool selection with specific LCA goals and integrating advancements in modeling and observational techniques to enhance agricultural sustainability.

The SG5b (**nitrous oxide**) methodological reviews highlighted that agricultural soils are a key source of N2O emissions which reflects in the livestock systems performance. Whilst LCA have been successful in assessing GHG from agricultural systems, no review and harmonization attempt has been focused on soil N2O emissions. The review therefore undertook a review and harmonization of existing methods to account for soil N2O emissions in LCA of agricultural systems and products, in relation to sources of N2O emissions including those originating from soils in relation to fertilisers (organic and mineral), crop residues, land use/land management change, grassland management, manure and slurry applications and from grazing animals. The review aimed to; i) to compare current methods used in LCA; ii) to identify advantages and iii) disadvantages of each method in LCA; iv) to suggest recommendations for LCA of agricultural systems; v) to identify research needs and potential methodological developments to account for soil N2O emissions in the LCA of agricultural systems. The approach adopted was based on two anonymous expert surveys and a series of expert workshops (number of workshop=21) to define general and specific criteria to review LCA methods for GHG emissions used in LCA of agricultural systems. A broad list of keywords and search criteria was used, and the reviewed papers and methodology were then assessed by LCA and soil N2O emission experts (n=14), which resulted in more than 25000 scientific papers and reports being identified. Of these, 1175 were screened and 31 scientific papers were related to soil N2O emissions.
The results showed that a high level of accuracy corresponded to a low level of applicability and vice versa. This highlights how the choice of LCA methods is critical for high quality agricultural system LCA, and should be based on the assessment objectives, data availability and expertise of the LCA practitioner. Whilst from an accuracy perspective, it is preferable to use a process-based model such as he DNDC model (after calibration and validation) or direct field measurements, considering system effects. However, when detailed data are lacking, the IPCC tier 2 methodology where available should be used, otherwise 2019 IPCC Tier 1 methodology. Further harmonization of methodologies is needed to improve the representation of agricultural management practices and soil-climate interactions, developing regression models that balance accuracy and applicability. However these models have limitations in the range of validity. A more integrated approach would refine emission factors and enhance the assessment of climate impacts from agricultural systems within LCA frameworks.

The SG5c (**manure and housing**) review for livestock LCA methods for livestock systems was successfully undertaken. It identified methods for GHG emissions focused on manure emissions and as with other categories, it was generally observed that a high level of accuracy corresponded a low level of applicability and vice versa. Thus, the choice of the methodology in relation to the LCA

objectives is particularly critical to enable high quality LCA assessments. Following the analysis of the available literature, a series of recommendations were proposed, and the choice of LCA methods should be based on the LCA objectives, data availability and expertise of the LCA practitioner. Whilst complex models have been developed for soil C and soil N2O emissions, for manure emission estimation, more complex emission factor equations have been conceived. Whilst IPCC Tier 1 methodology has been employed in most of the assessments analysed here, Tier 2 methods, related to the specifics of the manure and housing systems are preferable for improved accuracy. Independently of the method used, method limitations should be discussed in the LCA of livestock systems. This research provided a framework for potential improvements of the assessment methodology of manure management systems within IPCC categories.
Future development of LCA methodology is necessary to improve LCA of livestock systems, including the development of improved emission factors or preferably, basic process models which act as a compromise between applicability and accuracy. This LCA method development must be synchronous with improvements of observation methods and the assessment of different crop-livestock management.

The SG5d (**enteric**) methodological review found that the IPCC Tier 1 methodology to estimate methane emissions was employed in most of the assessments analysed. This simple estimation procedure limits accuracy, and where possible, and with suitable data availability, more complex methods should be adopted for greater accuracy, if appropriate input data and expertise are available. Recent studies have proposed more complex emission factor equations to improve enteric fermentation methane estimation.
Future development of LCA methodology is necessary to improve LCA of livestock systems. For enteric fermentation emissions, new inter-continental databases are providing improved accuracy using information from the intake of the animals and the composition of the diet. However, further research in developing a basic process model which results as a compromise between applicability and accuracy is desirable. Where emissions factors should better reflect herd characteristics and livestock management depending on the LCA objectives. This LCA method development must be synchronous with improvements of observation methods and the assessment of different crop-livestock management.

The SG6 (**Social LCA**) review found that whilst environmental considerations for agriculture have been widely explored, the social sustainability of agri-food systems has been scarcely addressed in literature. However, increasingly more focus is given to social impact assessment of both agrifood and livestock system in corporate reporting standards both at European and global level that also take account of extrinsic attributes connected with sustainability. Across the EU, the livestock sector plays a significant economic and social role, with European livestock farms employing around 4 million people, protein of animal origin covers over 50% of the total protein content of European diets. However, agriculture (including livestock) and forestry rank among Europe's riskiest professions due to frequent accidents and health concerns, jeopardizing sector sustainability. Yet, despite its size and impact on society, social life cycle assessment studies in agriculture and as well as in livestock production systems are very limited, therefore we undertook a review of S-LCA methodology for its application within the Pathways project.

Within the review we identified two main S-LCA methods, including the Reference Scale (RS) and Impact Pathway (IP) approaches. The RS S-LCIA uses an operational approach to describe a product system with a focus on its social performance or social risk and can be performed for all 40 impact subcategories, allowing for a broader scope of the study. This approach aligns with many S-LCA databases, and also enables the assessment of all stakeholder groups and their related impact categories, which makes them compatible with the multi-actor perspective. The IP S-LCA approach evaluates the outcomes stemming from the product system, including potential social impacts and employs one or more characterization models that utilize cause-effect relationships to evaluate impact categories. However, current development of characterization models within the IP S-LCIA is limited to potential social and socio-economic impacts for a single stakeholder category, mostly the workers, and for a very restricted number of impacts.

Whilst we found the RS method to be more developed and of wider application than IP, the literature also shows that some studies apply a mixed method including both RS-S-LCIA and IP Pathway approach, e.g., the emissions stemming from farm activities can be linked to human health, which could be explored.

Table 1. LCA recommendations for each key topics which have been reviewed and harmonized related to the LCA of livestock systems

Key Topic	LCA recommendations
Circular economy	• system expansion (without substitution) should adopted • Combine LCA with other multicriteria analysis assessments • Always include the environmental impacts of by-products and waste in the assessment
Biodiversity	• For biodiversity, it is recommended to characterise land-use intensity based on specific land management practices and include the positive impact of agroecological practices. • No methods reviewed at this stage is capable of properly assessed the impact on biodiversity related to land management change in a comprehensive and objective manner.
Animal welfare	• The Scherer et al., 2018 results as the most appropriate among the ones reviewed even though it presents some limitations.
Nutrition	• Among methods that were scored the highest, the SAIN,LIM and NRF are the most appropriate methods for including nutritional aspects in LCA on product level • when combining nutritional and environmental factors , we recommend considering impacts at a consumption level. • We recommend reporting the nutritional profile of the studied products in combination with the main results of environmental impact per nutritional functional unit. • We recommend considering nutritional guidelines and current food consumption patterns for selecting nutrients to be included in the nutritional scores. • We recommend approaching the combination of environmental and nutritional aspects in a diet context in future studies, where possible.
Soil CO_2 emissions	• To accurately assess soil C dynamics within a temperate climate, a time perspective of at least 20 is required. • a “spin-up” period is necessary for most models to simulate soi C dynamic. • Agroecosystem models such as DNDC or CropSys are preferred.

	• If less detailed input data is available, the IPCC 2019 Tier 2 steady-state methodology can be employed. • For broader, site-dependent or site-generic assessments, or when large-scale evaluations are needed, the use of Tier 2 methodologies such as the IPCC 2019 Tier 2 steady-state method or simplified carbon models like C-TOOL and ICBM is recommended (Andrén and Kätterer, 1997; Ogle et al., 2019; Petersen et al., 2013). • In cases of very limited information, or when data quality cannot be ensured or expertise is lacking, the IPCC Tier 1 methodology may be used (Ogle et al., 2019).
Soil N2O emissions	• it is preferable to use the DNDC model after calibration and validation or use of direct field measurements, taking in consideration system effects (Goglio et al., 2017). • when the necessary data are lacking, the use of IPCC tier 2 methodology (2019) with disaggregated EFs should be prioritized where available • Otherwise IPCC Tier 1 methodology following the 2019 guidelines should be used (Hergoualc'h et al., 2019). • When using 2019 IPCC Tier 2 or IPCC Tier 1 methodology to assess soil N2O emissions, the methodological limitations should be made clear by the LCA practitioner. • Independently from the methodological choice carried out, it is key to provide arguments for this choice and describe its potential limitations, in agreement with the ISO standards. • Especially for large site-dependent or site-generic studies (Potting and Hauschild, 2006), a preliminary assessment could still be carried out using simpler methods such as IPCC Tier 1 (2019) (Hergoualc'h et al., 2019). This should be complemented with a clear description of limitations of the methodology. Further, conclusions about these LCAs should be taken with caution as they poorly reflect local conditions and the effect of crop and grassland management.
Manure impact methodologies	• direct observations or a Tier 3 method are the most accurate and should be used in LCA of livestock ssytems, • In the absence of the necessary data, IPCC 2019 Tier 2 or the EEA 2019 Tier 2 methods are recommended (Amon et al., 2019; Gavrilova et al., 2019). • when applying an estimation method, limitations should be highlighted and discussed, especially if multiple methods are applied.

Enteric methane emissions	• for the purposes of an LCA, direct observations with specific devices or measurement within a metabolic chamber are preferable. • When these facilities are unavailable, it is recommended to apply the IPCC 2019 Tier 2 methodology for its wide applicability (Gavrilova et al., 2019). • Other equations can be applied that may be more specific to the feeding situation, e.g. based on Niu et al., (2018) for dairy cattle, Van Lingen et al. (2019) for beef cattle or for sheep (e.g. Belanche et al., (2023), • however, when non-IPCC methods are used, then limitations should be highlighted and discussed.
Social- LCA	• even though with some limitations, the representative scale social LCA approach could be used for the social LCA identified. • limitations of the methods should be always reported.

Introduction

This deliverable provides an overview of the methodological comparison of LCA methods undertaken though Task 5.1. Due to the wide focus of the review, the review was split into 6 topics. Whilst each maintained its own focus, the methods utilised were common across topics.

SG1 included aspects related to circularity within livestock LCA, including biomass utilisation.

SG2 assessed methodological aspects according to the focus area of biodiversity.

SG3 focussed its review on LCA methods associated with estimating animal welfare impacts.

SG4 reviewed methods to link nutritional quality aspects of food products to LCA.

SG5 included multiple GHG aspects linked to LCA within its remit including, SG5a (soil carbon), SG5b (nitrous oxide), SH5c (manure and housing) and 5d (enteric emissions).

SG6 reviewed current Social LCA (S-LCA) methods for their application to the Agri-Food sector.

The SG5b part of the report has been published in Agr Sys with the following reference: 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. Agr. Syst. 219, 104015. https://doi.org/10.1016/j.agsy.2024.104015.

SG1-3 and SG5a are at different stages of the publication process in several scientific journals.

SG1: Integrated environmental assessment of agricultural products: a screening review of handling circularity accounting in Life Cycle Assessment

Maxime Fossey[a], Pietro Goglio[b], Nina Röhrig[c], Klara van Mierlo[d], Annabel Oosterwijk[d], Alberto Maresca[e], Manuel Romero-Huelva[f], Simon Moakes[g], Marie Trydeman Knudsen[h], Camillo De Camillis[i], Lucia Rocchi[b], Laurence G. Smith[c,j], Nicholas M. Holden[k], Thomas Nemecek[l]

[a]Institut de l'élevage (IDELE), 149 rue de Bercy, 75012 Paris, France

[b] Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy.

[c]School of Agriculture Policy & Development, University of Reading, Whiteknights, RG6 6AH, UK

[d]Wageningen Social and Economic Research, Wageningen University and Research, Pr. Beatrixlaan 582-528, 2595 BM The Hague, The Netherlands

[e]SEGES Innovation P/S, Agro Food Park 15, 8200 Aarhus, Denmark

[f] Spanish Council for Scientific Research, Granada, Andalucia, Spain

[g] IBERS, Aberystwyth University, UK

[h]Department of Agroecology, Aarhus University, Blichers Allè 20, 8830 Tjele, Denmark

[i]Food and Agriculture Organization of the United Nations, Animal Production and Health Division, Rome, 00153, Italy

[j] Department of Biosystems and Technology, Swedish University of Agricultural Sciences, Box 190, SE-234 22 Lomma, Sweden

[k]School of Biosystems and Food Engineering, University College Dublin, Agriculture and Food Science Center, Belfield, Dublin 4, Ireland

[l]Agroscope, Life Cycle Assessment research group, 8046 Zurich, Switzerland

SG1 report is in preparation for submission in International Journal of Environmental Research.

Report's aims

The present report aims at providing a critical assessment of current LCA based methods used to evaluate the environmental impacts of the livestock sector. Covering the food, feed, fuel and biomaterial competition, crop-livestock interaction, and circular economy topics, this assessment, proposed adjustments to overcome the implementation gap CP have created for LCA, related to (i) their capacity to account for CP components and (ii) their consistency in terms of application.

This report provides in particular:

- An overview of LCA methodological frameworks used through the workshop management to cover all aspects related to this topic
- A descriptive analysis of the publishing dynamics on the topic
- A detailed analysis of the current methods/approaches with a set of criteria and evaluate their capacity to account for circularity in LCA
- A discussion focusing on two points: Multifunctionality and products/co-products/by-products (Definitions and openness to implications in allocations procedures
- A conclusion with recommendations for LCA of livestock systems

Abstract

Purpose

The agricultural sector is challenged to reduce its environmental impacts. These necessary changes are an opportunity to redefine functions of the livestock sector by moving its traditionally linear structure towards Circular Pathways (CP). Currently, common assumptions related to Life Cycle Assessment (LCA), predominantly method used to assess potential environmental impacts, only partially addresses CP. This review provides a critical assessment of current LCA based methods used to evaluate the environmental impacts of the livestock sector.

Methods

To frame the literature review, a participatory Delphi method was used to identify important agricultural and circularity concepts. An expert survey was carried out to select and rank both general criteria, used to evaluate the quality of the reviewed method regarding its relevance in LCA based approach, and specific criteria, used to evaluate the quality of the reviewed method regarding its

capacity to capture the CP concept. Then a structured review was carried out based on a set of common key search words to achieve a wide a coverage of the literature as possible.

Results and discussion

There is a need to improve and harmonise LCA methodologies to have greater coherence for LCA applied to the livestock sector. Encompassing circularity concepts such as (i) competition of use of products (food, feed, fuel and biomaterial use), (ii) closing nutrients cycles (crop-livestock interaction) and (iii) economic and social considerations (multi-functionality) in a single assessment still remains a challenge. The analysis showed multi-functionality and associated issues such as co-products, by-products and waste considerations as keystones in circularity in LCA. Areas of developments needed to reach better methodological compliance between level of accuracy and applicability were identified and several LCA approaches, such as combined approach covered themes not considered in standard LCA, seem relevant to capture CP dimensions.

Conclusions

LCA studies of agri-food systems may lead to conflicting conclusions about environmental impacts. Combined approaches need to be further developed to reduce variability. If these combined approaches may be considered more relevant to capture circularity in LCA approach, harmonization of assessment mechanisms through related common and normalized indicators is needed. This allows to improve the integration in LCA for a better consideration of circularity in environmental impact assessment.

Keywords
Agricultural products, Multi-functionality, Circular pathways, Environmental assessment, Life Cycle Analysis

Introduction

LIVESTOCK SECTOR AND ENVIRONMENTAL ASSESSMENT IN LCA

Globally, agriculture is a critical sector of the economy, providing food, feed and other resources that help sustain society. More particularly, the livestock sector contributes to (i) food security, nutrition and human health (9.2% of the world population faced chronic hunger in 2022) (FAO, 2023), (ii) both negative environmental impacts (the livestock sector is responsible for 14.5% of global greenhouse gas (GHG) emissions) (FAO, Gerber et al., 2013) and positive ecosystem services (habitat provision, landscape maintenance, source of nutrients and soil fertility maintenance, and carbon storage) (FAOSTAT; Oteros-Rozas et al., 2013) and (iii) economic growth with potential interdependencies regarding resource competition and circularity (TEEB, 2010; Stillitano et al., 2021; Oosting et al., 2021).

As it is important to minimize the adverse environmental impacts of agricultural production, it is equally important to consider expectations regarding the different uses of biomass and the reduction of resource consumption (Muscat et al., 2020). Almost 70% of the world's agricultural land is mobilized, directly (pasture and forage – 47%) or indirectly (concentrates and cereals – 23%), for livestock production (Steinfeld et al., 2006). However, considering the global livestock feed ration, 86% is composed of feed that is currently not edible by humans (FAOSTAT, 2016). Nevertheless, livestock still consumes about 30% of the global cereal production and up to 35% of current grasslands could be used as cropland (Mottet et al., 2017). Also considering that about 13% of cropland is used to produce biofuels and textiles (Poore and Nemececk, 2018), competition between agricultural land uses is still a major issue.

Considering the large land use of livestock systems, which is explained by demands for crops, grass and other feed (Van Zanten et al., 2018, 2022), livestock systems are at the heart of the competition between food, feed and fuel for biomass. Therefore, the concept of circularity, which is related to the reuse, upgrade and recycling of waste products aligning with previous research (Lindner et al., 2017). This in agricultural production could be one of biomass strategy management, which ensures sufficient production for both agricultural (feed) and human (food, energy) activities as initiated by the European Environment Agency (EEA, 2018), and reported by Muscat (2020). Thus, the role of livestock as converters of biomass in CP should be examined considering food security and nutrition, as well as environmental impacts.

Life Cycle Assessment (LCA) is widely used to assess potential (negative) environmental impacts of any stages of a product, process, or service (ISO, 2006). It typically focuses on negative impacts and often rarely considers positive contributions. The concept of life cycle benefit analysis is now emerging (Jones et al., 2022), but has little traction in the peer reviewed literature, perhaps because it offers scope for greenwashing? In the context of agriculture, positive outcomes might include greenhouse gas mitigation (Hererro et al., 2016), the provision of ecosystem services (Weiler et al., 2014) and the multi-functionality of agricultural activities (Notarnicola et al., 2017). Thus, by only partially considering the strengths and weaknesses of different agricultural systems, livestock environmental impacts can be misunderstood because a full picture of their contribution to sustainability in not available.

Addressing both multi-functionality and complex interactions between the livestock sector and other human activities, the concept of Circular Pathways (CP) has been suggested as a way to reduce food waste/losses impacts by first attributing biomass for human consumption and then utilizing by-products from the system. In this concept, livestock systems play an important role as converters of biomass not suitable for human consumption into food (van Zanten et al., 2018) and other raw materials (Nikodinovic-Runic et al., 2013). LCA has been predominantly used to assess environmental damage and to understand which pathways should be prioritized for certain types of agricultural waste by evaluating the implications of waste valorization within the CP (van Zanten et al., 2022). However, Talwar and Holden (2022) concluded that few LCA studies offer meaningful insight into sustainable transition of bioeconomy, not least because most of the studies reviewed in their paper did not consider the industrial symbiosis needed in order to properly understand the CP.

There is a need to improve and harmonize LCA methodologies that are capable of capturing circular aspects in agri-food studies in order to increase the consistency of results and reduce conflicting conclusions due to incompleteness. Indeed, assessments that focus on the success of an individual goal, such as food security, nutrition, agricultural productivity or environmental efficiency may miss the trade-offs that are important among these different goals (Obersteiner et al., 2016). Considering the multi-functionality of livestock systems through the CP concept within LCA approaches could lead to a reconsideration of the status of their inputs and outputs, in terms of resources consumption and waste disposal, and potentially could offer a more complete view of the environmental consequences of livestock systems.

CIRCULAR PATHWAYS FOR THE LIVESTOCK SECTOR IN LCA PERSPECTIVES

The idea of circularity in livestock systems contrasts with the common linear vision of agricultural production, which assumes that the system is an opened loop system without offset between the consumption of inputs (e.g., feed or fertilizer) and the provision of outputs (e.g., meat, milk, eggs) (Fig. 1).

With regards to livestock products, waste and systems, the CP concept is based on the idea of the "closing circle" (Commoner, 1971), which builds on two foundations: (i) the avoidance of use of arable land for producing animal feed that could potentially be directly consumed as human food (feed-food competition) or energy (feed-fuel competition), and (ii) the avoidance of waste generation (production efficiency) or the valorization of by-products (flow of biomass). Understanding of CP is continuously evolving (Borello et al., 2020), and the idea of closed loops (very efficient circulation of mass), is becoming relevant for the livestock sector as a framework for influencing behaviors and practices to minimize its negative impacts.

Livestock production is a major user of resources (e.g., water, mineral fertilizer, fuel, land), but it also generates flows of materials (e.g., manure, leather, wool, oils) valuable for several sectors such as the agri-food industry, textiles, energy, and industry. From a LCA perspective, interactions between systems should be identifiable beyond the farm scale by encompassing all material flows provided by livestock production and valued by another sector of activity (Fig. 1). Hybrid LCA offers a means of capturing inter-sector transaction that surround livestock but does not facilitate circular thinking.

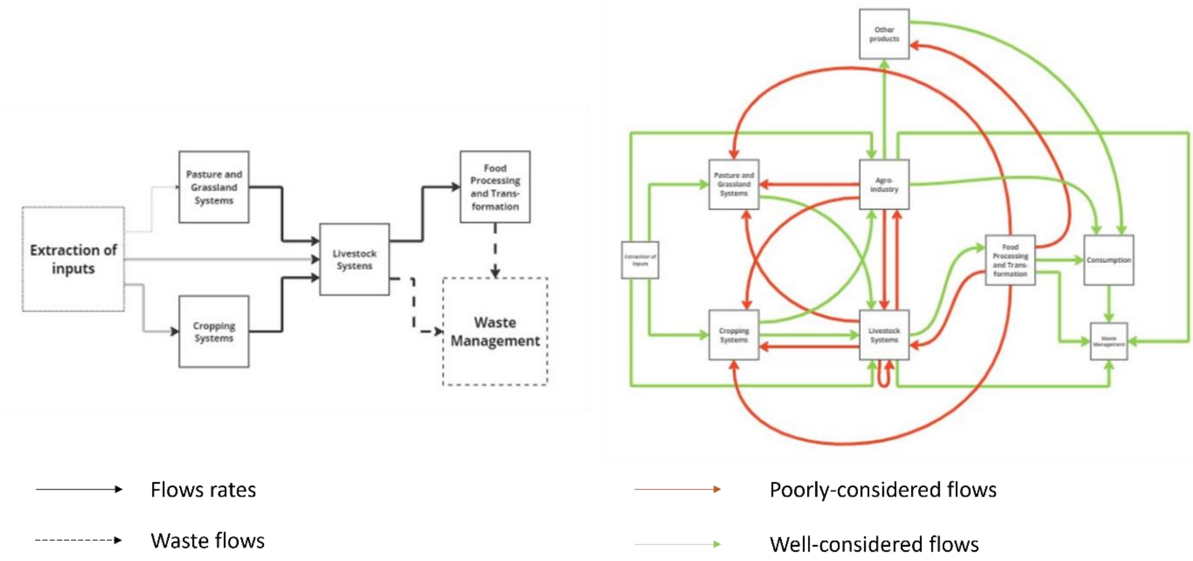


Figure 1 - Expectations for transition from a linear to a circular pathway adapted to livestock sector in LCA perspective.

Note: **Flows rates**: Flows are illustrated according to their intensity (thickness) and type (colour). Fertilisers and energy, as indirect flows are in grey, and animal feed from grasslands or cropping systems and associated practices impacts, and products provided by livestock systems (meat, milk, egg), as direct flows are in black. By-products and manure are in dotted lines; **Considered flows**: Circularity flows considered in LCA are illustrated according to their level of consideration. Green arrows illustrate the potential of well-considered flows in current LCA (e.g. energy, animal feed, livestock products, fields management). Red arrows illustrate the poorly considered flows in current LCA approaches (direct valorisation of by-products/co-products by agroindustry and the potential of recycling and valorisation of products provided by actors associated with livestock systems interactions.

In LCA studies of livestock production, system boundaries are often set to the farm gate, which restricts farming to the sole function of supplying food products and does account for its multifunctionality (Acosta-Alba et al., 2019) or the material flows valorization by other farms or other sectors (and Institute of Agriculture and Natural Resources, 2019). Thus, the complementarities and synergies between the livestock sector and other sectors can improve nutrient cycling and reduce resource consumption/competition by using non-arable land, crop residues and animal waste as fertilizers (Marton et al., 2016; Moraine et al., 2016). For example, about 19% of crop residues make up the global livestock feed ration (FAOSTAT, 2016) reducing the amount of waste generated by crop production. In addition, about 70% of the total nitrogen used for crop production is dedicated to livestock feed (Billen et al., 2014), and between 55% to 90% of ingested nitrogen and about 70% of phosphorus is excreted by livestock as urine and feces (typically described as animal 'waste') (Leip et al., 2019). These flows can be recovered as fertilizer to reduce resource consumption and GHG emissions from the manufacturing of mineral fertilizers (Moraine et al., 2016; Leip et al., 2019). Beyond these notions of transfers and recycling, integrated crop-livestock system (ICLS), by promoting this circularity, appears more resilient under changing climate context and could improve crop yield (Sekaran et al., 2021).

Animal waste is mainly considered at farm scale and applied as fertilizer and field amendments near or close to the farm. At a global scale, this could represent about 30% of total nitrogen inputs and 50% of total phosphorus (Dourmad et al., 2019 – data from French system). Thus, the use of livestock waste for energy recovery through anaerobic digestion is very important and should be associated with fertilizer production. In addition, livestock waste and co-products may also be considered beyond the farm gate, including wastes from the transformation stages of the value chain associated with livestock farming (e.g., offal, leather, etc). While material (nutrient) and energy recovery (via anaerobic digestion) of manure are generally considered (Leip et al., 2019; Awasthi et al., 2019), valorization of by-products of the agro-food industries, and as new sources of animal feed is constantly under investigation (Campos et al., 2019; Costantini et al., 2020; Al-Zohairi et al, 2022). Thus, a better understanding and knowledge of by-products valorization and recovery of wastes from the livestock sector, by developing new sources of animal feed or improving the

management of by-products in manufacturing processes, is relevant to the CP approach (Siddiqui et al., 2021).
CP also stimulates a rethinking of the definition of 'waste', with terms such a co-products and by-products, avoidable and unavoidable, or food and residues now being used. This might have implications regarding the allocation procedures between products (ISO, 2006; Curran, 2008; de Vries and de Boer, 2010), which is a key methodological issue in LCA, and could change understanding of the environmental consequences of livestock systems.
Regarding the livestock LCA, three issues need to be examined in greater detail in order to understand the implications of using LCA in a CP context: (i) competition between production of food, feed, fuel, and bioeconomy feedstocks, either as alternatives (substitution of one product by another one) or adversaries (seeking the same limited resources as inputs, (ii) interactions between production systems (efficiency and/or intensity of flows between systems) especially between crops and livestock, and (iii) multifunctionality of the livestock system through better understanding of all products and services that is can provide beyond the function of supplying food.

Methodology

LITERATURE REVIEW PROCEDURE

To provide the framing for the literature review, a participatory Delphi method was used as a series of structured surveys with different stakeholders (Mullender et al., 2020), to identify important agricultural and circularity concepts. Initially an anonymous survey using Google Survey (Google, 2021) was used to allow LCA experts to propose and rank terms, which were analyzed to identify five important criteria (Rosner, 2011) related to "circularity in the livestock sector". Subsequently an expert survey to select and rank criteria was also carried out following the approach of Mullender et al. (2020). The general criteria were drawn from RACER (Wideman et al., 2009), JRC (JRC, 2010; Zampori and Pant, 2019), LEAP (FAO, 2018) and definitions from the Association de Coordination Technique Agricole, ACTA (the French farmer' development board association). Both datasets were further screened during LCA expert discussions (29 workshops with 21 different experts in livestock and LCA, drawn from academia, agriculture and farm advisory boards (Goglio et al., 2023)) to provide a set of framing concepts for making sense of the literature review of the LCA methodology for livestock systems and products (supplementary materials, Table 1). Finally, a set of specific evaluation criteria were agreed during 4 workshops for a community of expert peers. The definition and the scale of specific criteria were reformulated and modified to ensure rigor and coherence in the analysis of the LCA methods. At the end of the process a list of criteria and means of interpreting them were available to structure the analysis of the literature (supplementary materials, Table 2).

The process of developing both general and specific criteria is described in detail in Goglio et al. (2023). While general criteria are used to evaluate the quality of the reviewed method regarding its relevance in LCA based approach, specific criteria are used to evaluate the quality of the reviewed method regarding its capacity to capture the CP concept. There are three or four levels for each criterion, where levels 1, 2, 3, and 4 correspond to scores 1, 2, 3, and 4, respectively (supplementary material, Table 1, Table 2). Lower scores represent the worst performance, and higher scores represent the better performance.

A structured review (Figure 2) was carried out searching Web of Science, Scopus, and Google Scholar databases and selected grey literature such as FAO LEAP reports and the PEFCR general guidelines (FAO, 2018, 2020; Zampori & Pant, 2019). A set of common key search words (Table 1) was selected to achieve high initial recall to ensure a wide a coverage of the literature as possible. The search strings included terms related to LCA, circularity, livestock systems, animal products, alternative and interactions (e.g., food-feed-fuel).

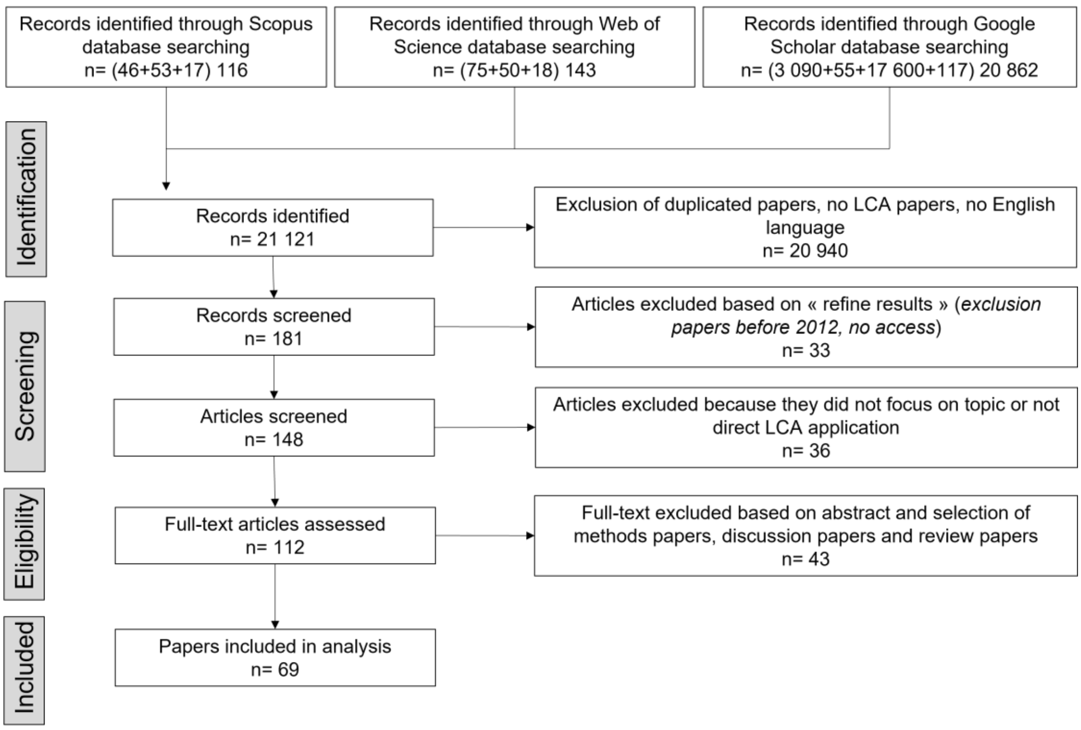


Figure 2 - Methodological steps of the literature search process

Database	Combination	Search strings[1]
Scopus & Web of Sciences	1	("LCA" OR "Life cycle assessment" OR "Life cycle analysis) AND ("Circular food system" OR "Circular economy") AND ("livestock" OR "dairy" OR "cattle" OR "sheep" OR "pig*" OR "poultry" OR "goat*" OR "milk" OR "egg*" OR "chicken*" OR "cow*")
	2	("LCA" OR "Life cycle assessment" OR "Life cycle analysis") AND ("feed-food" OR "feed-fuel" OR "food-feed" OR "food-fuel" OR "fuel-food" OR "fuel-feed")
	3	("LCA" OR "Life cycle assessment" OR "Life cycle analysis") AND ("crop-livestock" OR "crop/livestock") AND ("integrat*" OR "interaction*" OR "mixed system*")
Google Scholar	1	LCA OR "life cycle assessment" AND "circular economy« AND animal OR livestock
	2	LCA OR "life cycle assessment" AND "feed/food" OR "food/feed" AND fuel OR feed OR food
	3	LCA OR "life cycle assessment" AND feed AND substitute OR alternative
	4	LCA OR "life cycle assessment" AND "crop/livestock« AND interaction

[1] *Last accessed on 07 March 2022*

Table 2 Query used in database searching

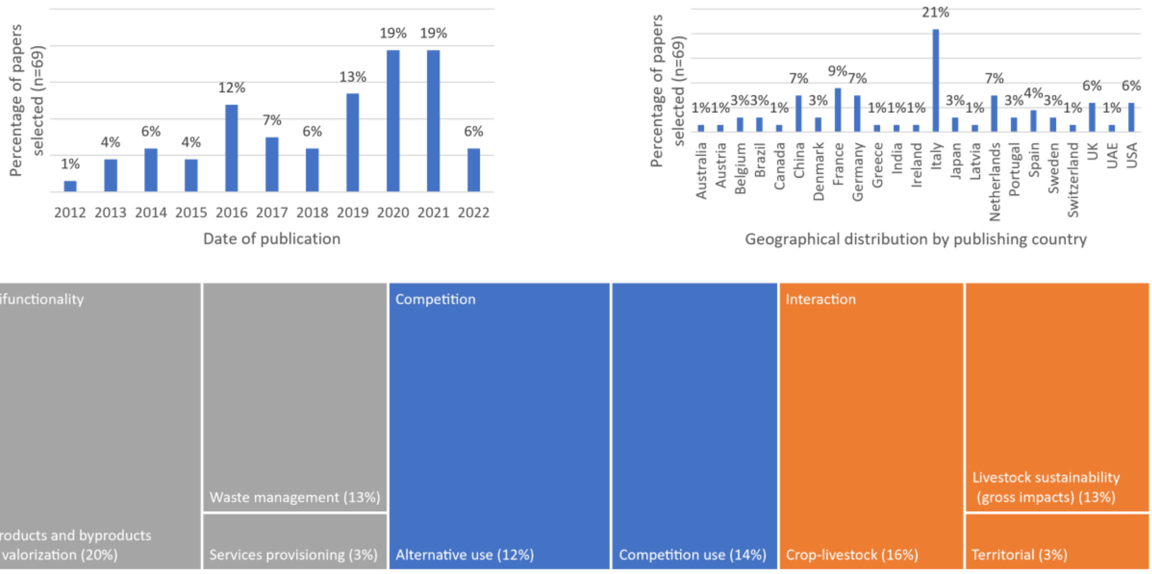
The initial search yielded 21,121 documents. Screening for duplicates, English language and LCA-focus in the abstract reduced this to 181 documents. Subsequently restricting to documents from 2012 to March 2022 and public availability (both open access or behind a paywall) reduce the set to 148 documents. Screening the full text left 112 papers and a final assessment to eliminate methodology papers, review and conceptual discussions left 69 documents to be analyzed using the framework established by the participatory Delphi method.

Results

DESCRIPTIVE ANALYSIS OF IDENTIFIED PAPERS

More of the selected papers were published in the latter half of the period 2012 to 2022 (Figure 3a). Most of the publication were of European origin (71%), followed by China (7%), USA and UK (6%) (Figure 3b). For European Union countries, the five highest-ranked origins were Italy (21%), France (9%), Germany and Netherlands (7%) and UK (6%).

Based on the CP framing (supplementary material, Table 2), the three dimensions were equally embedded in LCA approaches, with addressing multifunctionality, 33% competition and 31% interaction (Figure 3c). The competition dimension largely referred to trade-offs between the expectations of agriculture to provide food and energy security, rates of consumption of edible food and land use requirements. Alternative uses were addressed in 12% of the documents while adversary featured in 14%. The interaction dimension focuses on the "crop-livestock interaction" issue (16%), which included Integrated Crop Livestock Systems (ICLS) to reduce fertilizers use, energy demand, and system organization. Management strategies were the focus of 13% of the studies, mainly using attributional LCA, and 3% related to off-farm impact and spatialized LCA approaches. The main objectives of these studies were to assess the influence of the spatial organization of farms on the environmental and to develop emissions factors at territorial scale. The multifunctionality referred to supplying additional products and functions from livestock (Acosta-Alba et al., 2019; Weiler et al., 2014). Only 3% of the papers dealt with the service provision function of livestock, but 20% addressed valorization of co-products or by-products (e.g., use and recycling of agricultural outputs), which increased to 33% when used of animal waste was included. The main focus was allocation methods and data.



Circular Pathways dimensions and main issues

Figure 3 - Publication trend by (a) year, (b) country, and (c) main issues encountered by Circular Pathways dimensions (Paper search ended on 7 March 2022)

LCA APPROACHES TO CIRCULARITY

The most common LCA approach to circularity, adopted by 65% of documents, was attributional LCA. These studies were used several impact methods aligned to ISO 14044 and 14046 representing up to 16 different impact categories. Documents mentioned the International Reference Life Cycle Data System (ILCD) (51%), (CML (36%)), IMPACT (27%), ReCiPe (23%) and the cumulative energy demand (CED) (10%) when describing impact methods. Almost 28% of papers focused on GHG emissions using IPCC guidelines.

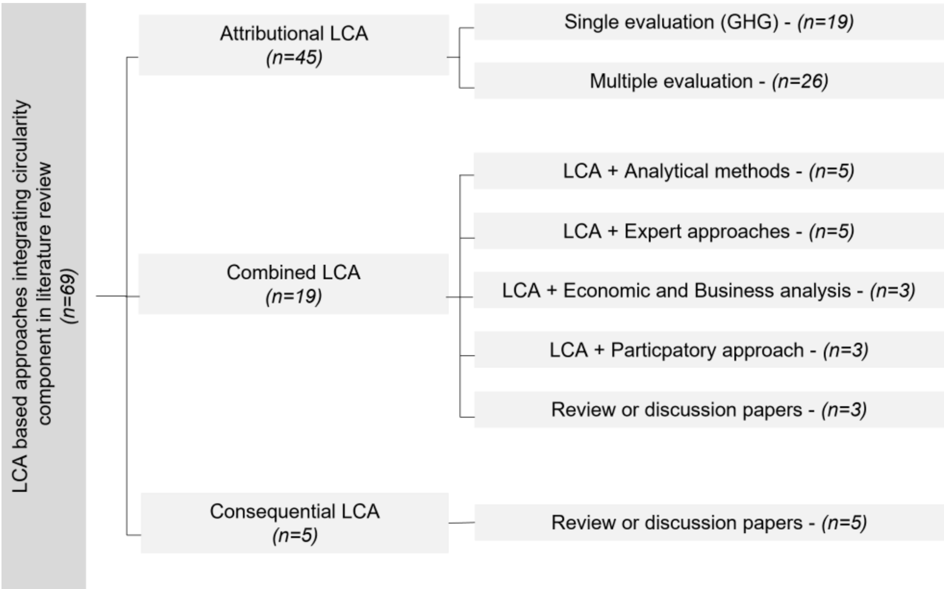


Figure 4: The main LCA based approaches to considering circularity in livestock sector

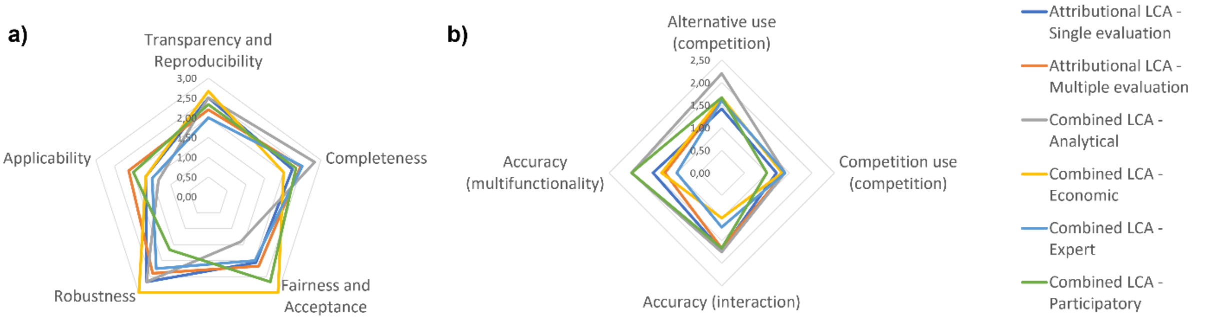
Studies in which LCA was used in combination with other approaches, grouped under a "Combined LCA" category (27%), were categorized as: (i) analytical-based (31%) such as Material Flow Analysis (MFA), Data Envelopment Analysis (DEA) or Nexus Approach, (ii) expert-based (21%) such as coupling LCA with Emergy calculation, or agriculture-specific allocation proposal, (iii) Economic (16%) such as the "choice of technology model", defining by-products as coproducts or waste based on technological and economic capacity criteria, or the circular model approach of Bech et al. (2019) which pairs LCA to a product/service-systems business model, and (iv) social-based (16%) such as participatory approaches accounting for the multi-functionality of agriculture. These combined approaches covered themes not considered in standard LCA such as energy (exergy and emergy), service system by decoupling value creation from resource consumption or social functions. In addition to these approaches, to address both data availability and subjective assumptions related to the operator, some digital tools (i.e., mathematical and linear programming optimization models) were integrated to aid decision-making.

Consequential LCA was used in only 7% of the documents with a focus on Land Use Change (LUC) or the impact of using residual biomass as a resource for valorization. For instance, in exploring the environmental consequences of using waste-fed-larvae meal as a feed ingredient.

COMPARATIVE ANALYSIS AND VARIABILITY BETWEEN LCA BASED METHODS

The analysis of the scores assigned to each LCA approach and each dimension of the CP framework (figure 5; supplementary material, Table 2) reveals how each method and approach captures the concept of circularity and the issues identified as important by experts. Based on general criteria (Figure 5a), the combined LCA with an economic approach scored highest (2.47), while expert approaches scored lowest (2.05). Looking at individual criteria, the combined LCA-analytical approach was not satisfactory for applicability and fairness and acceptance while the combined LCA-expert appears not satisfactory only on the applicability criteria. These low scores are explained by the fact that both approaches have less consensus for consistency and comparability.

Furthermore, variability between approaches appears greatest for fairness and acceptance and robustness, and lowest for transparency and reproducibility and completeness. While combined LCA-Analytical is perceived as the least fair, mainly due to the inconsistent application of the method to all data and assessment stages, combined LCA – Participatory was scored least robust, due to the deviations observed from the recommended LCA guidelines in terms of allocation or functional unit. Otherwise, both transparency and reproducibility and completeness criteria were relatively homogenous reflecting the good coverage of flows included in the inventories and the availability of documentation on which approaches are based on.



	Transparency and Reproducibility	Completeness	Fairness and Acceptance	Robustness	Applicability
Attributional LCA - Single evaluation	2,50	2,24	2,06	2,67	1,67
Attributional LCA - Multiple evaluation	2,20	2,46	2,08	2,40	2,13
Combined LCA - Analytical	2,60	2,80	1,40	2,60	1,20
Combined LCA - Economic	2,67	2,00	2,33	3,00	1,67
Combined LCA - Expert	2,00	2,60	1,80	2,40	1,60
Combined LCA - Participatory	2,33	2,33	2,67	1,67	2,00

Higher ranking · Lower ranking

	Alternative uses (competition)	Accuracy (competition)	Accuracy (interaction)	Accuracy (multi-functionality)
Attributional LCA - Single evaluation	1,42	1,21	1,72	1,53
Attributional LCA - Multiple evaluation	1,62	1,38	1,67	1,26
Combined LCA - Analytical	2,20	1,40	1,75	2,00
Combined LCA - Economic	1,67	1,33	1,00	1,33
Combined LCA - Expert	1,60	1,40	1,20	1,00
Combined LCA - Participatory	1,67	1,00	1,67	2,00

Not satisfactory

Figure 5 - Results of general (a) and specific (b) criteria scoring considering LCA based approaches

*Note: **Attributional LCA**: Common LCA approach according to ISO 14044 and 14046 and including indicators representing one (single evaluation) or several indicators (multiple evaluation) among the 16 different impact categories; **Combined LCA**: Common attributional LCA approach for which either an impact method used differs* *from ISO recommendations, or for which flows not initially considered are added using analytical, economic or social (participatory) approaches*

Discussion

This screening review, integrating competition and interaction between products, production systems, and multi-functionality, addressed crucial issues related to the livestock sector, e.g., European interest in waste (the Waste Management Directive; European parliament, 2018), and the newer livestock systems (e.g., ICLS, mixed) that have emerged to address the negative externalities of farming (Duru and Therond, 2021; Sekaran et al., 2021). These technical levers are part of the strategy promoted by the European Union (Green Deal, 2019) to reduce GHG emissions by 55% by 2030 and to achieve climate neutrality by 2050 and are considered in greater detail below. Generally speaking, European papers dominated the results, which was also reported by Esposito et al. (2020) and Stillitano et al. (2021). This reflected the emphasis that the European Union placed on the development of the sustainable performance of food products (European Commission, 2015; Petit et al., 2018). The large number of associated scientific papers in this area (Figure 3) and the heterogeneity of results observed through this review highlights the need for a harmonized methodology. Indeed, sustainability assessments of a wide range of agricultural products and comparison at European scale (at least), does not seem compatible with having multiple different methods used by researchers and practitioners. The lack of agreed impact assessment methods, reflecting the "sustainability dimensions" of the body of work reviewed raises the question of whether each dimension is perceived as equally important, an issue that is open to academic debate (Stillitano et al., 2021).

STRENGTHS AND LIMITATIONS OF LCA APPROACHES AND PROPOSED DEVELOPMENTS

Based on these results and following the general LCA methodological recommendations provided by FAO LEAP guidelines (2016,2018,2020), there are four major decisions made by the practitioner that can affect LCA: the system boundary, where data come from and the quality of data required, and the functional unit and how multifunctionality is handled.

In general, most of the papers reviewed are based on attributional LCA methodologies conducing to evaluate the potential environmental impacts associated with a product system using characterization factors in line with common guidelines (IPCC, ILCD, FAO LEAP). This finding can be explained by the fact that the studies seek overall to obtain comparable results making the above approaches the most suitable. Indeed, the lower variability observed in scoring for these approaches

(-30% of variability in comparison with other approaches) reflects a relative agreement ensuring the capacity to compare results independently of studies what can be sought for environmental claims of agricultural products, but which are not an objective of the ISO rules (McLaren et al., 2021) In addition, while combined LCA approaches could appear as more accurate for capturing specific criteria related to circularity (**Error! Reference source not found.** – average higher score of 4%), the limiting ability to use by practitioners and consecutive dissemination (average lower score of applicability of 15%) reflect the need for development and harmonization for these alternative approaches. In the same way as for combined approaches, if consequential LCA seems to be relevant to account for benefits and drawbacks of livestock systems at wider scale than farm (plant or territorial scale) and assessing interactions across system alternatives to a particular product or service (Schaubroeck et al., 2021), it is still remain open questions about market assumptions, geographical specificity and value judgments for interpretation (Ward et al., 2016) and uncertainties brought by complex models of economic mechanisms and market predictions (Finkbeiner et al., 2014).

To date, capturing circularity related to livestock systems through LCA is still challenging. Nevertheless, this review shows that if it is possible to reach a relatively satisfactory accounting for these needs in LCA based approaches, these latter induce some overlaps between allocations rules, functional unit definition and boundary of studies. However, to avoid the over-iterative processes necessary for the current assessment of circularity (step-by-step assessment of assumptions), some approaches and methods of this review seem relevant regarding the CP dimensions.

If compiling an inventory of relevant input and output of a product system does not appear to be limiting in that the data are increasingly available, it is the consideration of concepts related to circularity such as (i) competition of use of products (food, feed, fuel and biomaterial use), (ii) closing nutrients cycles (crop-livestock interaction) and (iii) economic and social considerations (multi-functionality) in a single assessment that remains a challenge as discussed below.

COMPETITION AS AN ABILITY TO CONSIDER POTENTIAL ALTERNATIVE USES

In these analysis frameworks, mainly functional unit, allocation and boundaries issues are assessed, since they are mutually interdependent and any change on one cannot be made without considering the adaptation of the others. First, while by default, the functional unit is per mass or per volume, if the study justifies other objectives than compare products sold, additional functional unit can be selected, as recommended in LCA of cropping systems (Goglio et al., 2017; Nemecek et al., 2011). Here, to capture circularity, several functional units based on surface (Berton et al., 2020), energy (Ghisellini et al., 2015), nutrition requirements (Mottet et al., 2017; van Boxmeer et al., 2021) have been proposed to capture competition issues such as land, fuel and feed-food respectively.

While the combined LCA – analytical approach is highest scored for the alternative uses issue, with Data Envelopment analysis (DEA) (Laso et al., 2018) with linear programming methods, it is the combined LCA -Expert approach which obtained the higher score, with Material Flow Allocation (Ghisellini et al., 2015), RCOT (Rectangular Choice-of-Technology) (Springer and Schmitt, 2018) or business model. (Bech et al., 2019) for the competition use.

The DEA methodology consists in deriving objective weights using linear programming tools without making any previous assumption on data and using existing quantitative data on different stages of the life cycle (Cooper et al., 2007). This method is an analytical method for measuring the efficiency of a system or economic unit by comparing its inputs and outputs. It seeks to determine the best use of available resources by identifying the solutions that achieve the highest ecoefficiency. In the context of the LCA, the DEA method can be used to evaluate different alternatives to use a product, based on several criteria such as energy efficiency, greenhouse gas emissions, water consumption, etc. For example, it can be used to evaluate which product use option offers the best resource efficiency and the lowest environmental footprint.

The Material Flow Allocation Model analyzes the material flows between the different uses of agricultural resources (food, feed, biofuels) and assigns the environmental impacts associated with each use (Ghisellini et al., 2015. This makes it possible to compare the environmental impacts of different uses and determine their competitiveness. The multi-objective optimization models, such as RCOT (Duchin and Levine, 2011) or business, analyze the different competing objectives (for example, food security, energy security, reduction of greenhouse gas emissions) and tries to find optimal solutions that simultaneously maximize these objectives. It can be used to identify trade-offs and synergies between the different feed, food and fuel uses. These solutions could then be evaluated and compared to select the one that offers the best overall compromise between environmental and economic performance.

By using these models in a LCA, decision makers can make informed decisions about product use alternatives or competition, considering both efficiency and environmental impact. This can help identify best practices and opportunities for improvement to reduce the environmental footprint of products and promote more sustainable use of resources. However, according to the example of the GREET model (Hoekman et al., 2017), it may be necessary to develop more sectorial full life cycle as database in order to obtain all the data necessary for the proper use of these models.

INTERACTION BETWEEN THE LIVESTOCK SECTOR AND OTHER HUMAN ACTIVITIES

Another component observed is the economic approach with business models as environmental performances optimization. In Bech et al. (2019), a circular business model integrated to LCA assessment allows to avoid misinterpretation of standard environmental results by correcting impact values considering the closed loop recycling. The question of recycling and the reuse of products or by-products is also addressed in Springer and Schmitt (2018) in which technological and economic capacity is taken into account to distinguish co-products from waste. Thus, according to Springer and Schmitt (2018), since it seems more impactful to recycle a co-product than use a "new resource" as raw material, this co-product could reasonably be considered waste. These kinds of considerations provide an explicit methodological approach to relate CP and sustainability and provide a practical support for decision-making process in collaboration strategies improving environmental performance while reducing resource consumption.

For this CP dimension, both the combined LCA-analytical and expert approaches with the DEA method and business model (Bech et al., 2019) respectively appears of interest. While they present some strengths and limitations as described above, an additional list of by-products potentially usable as raw material for human activities may be also required to allow the development of specific life cycle inventory.

MULTI-FUNCTIONALITY OF LIVESTOCK SYSTEMS

Functional units, allocations or boundaries are mutually interdependent. In the study Houssard et al. (2020) and Marton et al. (2016), a system expansion approach was proposed to optimize the use of milk components or to better assess the multi-functionality of the system, in compliance with circularity principles as key to reduce environmental burdens of the food system. However, the combined LCA-participatory is the highest scored approach to define all products/services provided by livestock. If this approach could appear subjective and context-dependent, resulting in multiple no-comparable items, a guideline defining the main functionalities of agricultural sector considering the capacity of the system to produce food, energy, biomaterial, maintaining landscape or biodiversity quality and the rules to apply for taking into account (allocation, perimeter, etc ...) could provide a complementary basis for integration within existing guidelines.

MULTI-FUNCTIONALITY, CO-PRODUCTS, BY-PRODUCTS AND WASTE - KEYSTONES IN CIRCULARITY IN LCA

Papers developing "combined LCA" aimed to encompass agricultural challenges through the integration of alternative multi-criteria methods. Among these approaches, it was observed that participatory, social research conducted with farmers (Acosta-Alba et al., 2019) was used to define the multi-functionality of livestock systems. Allocation of environmental burden based on farmer assessment of farm function (Weiler et al., 2014) allowed the integration of the social value of livestock to go beyond the hierarchy of allocation rules provided by LCA guidelines (FAO 2016). These considerations of multi-functionality and site-specific characterization could provide important information on how environmental impact for is calculated for farms and systems of different sizes, but could create a 'collective subjectivity' that might drive methodological inconsistency through time and space.

This review revealed that co-products and by-products (when further use is already defined) were usually treated using allocation rules (ca. 50% of papers use mass or economic allocation rule) before the system expansion (without substitution) rules (ca. 20% of papers). While these methods correspond to the common procedures recommended by ISO 14044 (ISO, 2006a, 2006b) and recommended by FAO LEAP guidelines (FAO, 2020), the preferred use of allocation method (last priority of ISO standard recommendation after system subdivision and expansion) reflects the complexity of dividing environmental impacts between the product and their potential co-products and by-products. Considering livestock system, more than one product is produced. Therefore, the system expansion rule (without substitution), by integrating both co-products and/or by-products may be preferred to allocation rules to be more coherent with ISO standard priorities rules.

To go further in this way, methods corresponding to a process mapping step both recommended by ISO 14001 (ISO, 2015) and PAS 2050 (BSI, 2011) could be applied (around 30% of papers). This approach avoids the use of allocation rules but requires to re-define the usual mass functional unit that lead to better consider the multi-functionality of livestock systems. As reported in Acosta-Alba et al. (2019), a functional unit only based on the mass of the main products sold may "restrict farming systems to the sole function of supplying food products and does not correspond to the reality of multifunctionality", as widely previously discussed for agricultural systems (Nemecek et al., 2011; Goglio et al., 2017). Often resulting from social approaches, alternatives functional units could refer to landscape or social services such as production per cultivated area or livestock production per unit of forage area (Acosta-Alba et al., 2019). It may better reflect whether adaptation/environmental resilience (Payraudeau and van der Werf, 2005) by referring to the use of consumable resources (feed, food or biomass) related to the intensity of emissions by area or by services produced.

Focusing on material goods (i.e., exclusion of landscape or social services), the use of the system expansion (without substitution) rules requires to recognize the potential value of residues by considering upstream and downstream impacts and require adaptation to avoid misguided decisions for a low-impact circular pathway, as argued by Olofsson and Börjesson (2018). These adaptations go through two stages: (i) a clear definition of "waste materials" and (ii) an alternative to zero burden assumption. First, the European Union's Waste Framework Directive (WFD - 2008/98/EC) defined waste as "any substance or object which the holder discards or intends or is required to discard". The concepts of "co-product and by-product" are defined as "material that are not the principal material product" (product) but material produced as an integral part of the main production process. While co-product can be defined by a costing value as it is already corresponding to a specific use in a defined market, a by-product is not a deliberate material. This by-product is then considered as waste if no "further use" is certain (WFD, 2008). Therefore, all by-products that allow financial gain or integrate into a solid market (supply and demand) shall not be considered as waste but as co-product (if no further processing is requested) or as primary material (if further processing is required) and considering for upstream impacts and thus avoiding the associated zero-burden assumption. Thus, as reported by Pelletier et al. (2015), the waste definition based on this "further use" appears consistent with the ISO standard (ISO, 2006) and a systematic consideration for "residual" material in LCA could be proposed based on this definition as illustrated for wasted food and food residue (Oldfield et al., 2016). Therefore, a systematic estimate of the feasibility of using waste could applied to the allocation rules defined for co-products.

Considering by-products (with no further use) and wastes, Springer and Schmitt (2018) proposed a method to "define by-product as co-products or waste depending on the technological and economic capacity to utilize them". This approach of residue valorization capacity could avoid "both misconceptions of them as per default environmentally preferable resources, and unintentional support for high-impact primary production systems" as reported by Olofsson and Börjesson (2018) and also entails a real low-impact circular economy. Therefore, if wastes and by-products "further use" can be defined based on (i) substitution capacity of primary production and (ii) the relevance of using by-products and wastes as an alternative considering both environmental and economic costs of its production (further process), they could be "easily" defined as effective waste or potential co-products. Thus defining, these materials could be integrated, or not, in a life cycle inventory of a product with impacts according to ISO standard (allocation procedure) as suggested by Laurenti et al. (2017).

These considerations could be seen as crucial noting that waste issues appear as the keystone in these circularity approaches. Indeed, both direct waste recovery (manure at farm scale), or indirect valorization of by-products of the agri-food industries (at factory scale) or co-products of other activities (at territorial scale) for alternative feeds are explored. For these purposes, some technologies of valorization of agricultural waste (mainly anaerobic digestion - Awasthi et al., 2019, Diaz et al. 2021) or some technologies emphasizing through new sources of animal feed (Zanten et al., 2018, Campos et al., 2019)) or new manufacturing processes to handle by-products are performed and assessed (Antoniadou et al., 2020, Schestak et al., 2022). Considering that these technologies could have both benefits and drawbacks at different scales, it seems crucial to consider both farm and territorial scales, especially in terms of energy competition and "avoid products" issues. Moreover, taking into account of these processes within LCA approaches implies strong assumptions (define potential use and assess the technical feasibility in environmental way), including an alternative to zero-burden assumptions for waste, and related consequences both in terms of methods and results. Thus, it appears that the product scope alone is insufficient where the objective is to assess the multi-functionality of the system and needs to be expanded to consider the whole food system. This expansion would extend the sole function of the product to the surrounding services and thus defer the impacts to all these services.

Conclusion

The agricultural sector faces new challenges in which resource efficiency, environmental neutrality, and societal acceptance with reference to food and energy security, is demanding for innovating technologies and methodological development to assess circularity approaches in livestock value chains and systems. These developments are opportunities to consider multi-functionality and environmental benefits of livestock systems and promoting the transition towards more sustainable performance of the latter.

According to this review results, integrating and well as assessing the CP concept in LCA tools is beneficial for the agricultural sector. While system efficiency (minimization of both upstream inputs and downstream residues) is still the major component and probably should stay as a basis of environmental assessment, the "circular efficiency", considering the capacity to reuse by-products, leads towards more sustainable livestock systems. Therefore, moving away from the attributional LCA approach, beyond the contribution to reduce environmental impacts of livestock systems that this seems to imply, allows support for a more holistic analysis, to understand which pathways could be considered for a given livestock system in a given territory. The main limitations of the development of such tools are the assumptions (especially economic with unstable market values) that must be made (both consequential and combined approaches) and the availability and level detail on data if spatialization is considered.

In conclusion, livestock systems consume, process and provide large quantities of resources and bioresources. Diversity of systems, practices and resources consumed and provided by livestock systems is a major advantage for CE but it appears that a need for better assessment of flows through relevant indicators is required to be better captured in LCA. "Combined approaches" aiming to provide a complete assessment and analysis of livestock systems are crucial in the context of comparison of agricultural products and environmental labelling because of they could provide an insight of potential environmental benefits, trade-offs and disadvantages. These should be recommended for a comprehensive assessment of benefits, trade-offs and disadvantage of agricultural systems, products, by-products and waste management. Thus, input and output flows at unit production needed to assess the impacts of agricultural production through common indicators in line with standards guidelines are relatively well considered within LCAs and appears in the majority of the screened papers due to ease of use, comparison and understanding by LCA practitioners. Therefore, the challenge of integrating circularity into LCAs lies in understanding the multi-functionality of production systems and scales of interaction throughout the product value chain. This apprehension of multi-functionality can find answers in the proposed combined approaches, integrating social and economic dimensions in addition to environmental interactions. These dimensions lead us to reflect both on the functional units best adapted to agricultural production according to production systems and on the procedures of allocations which must be accompanied by more framed definitions of the co-products, by-products and waste.

Combined approaches, need to be further developed in a collaborative way with the aim to reduce variability observed by the production of standardized indicators. If these combined approaches may be considered more relevant to capture circularity in LCA approach, harmonization of assessment mechanisms through related common and normalized indicators is needed to improve the integration in LCA, for a better consideration of circularity in environmental impact assessment.

References

Acosta-Alba, I., Chia, E., Andrieu N., 2019. The lca4csa framework: Using life cycle assessment to strengthen environmental Acosta-Alba, I., Chia, E., Andrieu N., 2019. The lca4csa framework: Using life cycle assessment to strengthen environmental sustainability analysis of climate smart agriculture options at farm and crop system levels. Agric. Syst., Elsevier Masson, 2019, 171, pp.155-170. https://doi.org/10.1016/j.agsy.2019.02.001

Adghim, M., Abdallah, M., Saad, S., Shanableh, A., Sartaj M., Eltigani El Mansouri, A., 2020. Comparative life cycle assessment of anaerobic co-digestion for dairy waste management in large-scale farms, J. Clean. Prod., Volume 256,120320, ISSN 0959-6526. https://doi.org/10.1016/j.jclepro.2020.120320.

Al-Zohairi, S., Knudsen, M.T., Mogensen, L., 2022. Environmental impact of Danish pork-Effect of allocation methods at slaughtering stage. Int J Life Cycle Assess, 27 :1228-1248. https://doi.org/10.1007/s11367-022-02089-y

Aoki-Suzuki, C., Dente, S.M.R., Tanaka, D., Kayo, C., Murakami, S., Fujii, C., … Hashimoto, S., 2021. Total environmental impacts of Japanese material production. J Ind Ecol, 1-12. https://doi.org/10.1111/jiec.13152

Awasthi, M.K., Sarsaiya, S., Wainaina, S., Rajendran, K., Kumar, S., Quan, W., … Taherzadeh, M.J., 2019. A critical review of organic manure biorefinery models toward sustainable circular bioeconomy: Technological challenges, advancements, innovations, and future perspectives, Renew Sustain Energy Rev, Volume 111, Pages 115-131, ISSN 1364-0321. https://doi.org/10.1016/j.rser.2019.05.017.

Bech, N.M.; Birkved, M.; Charnley, F.; Laumann Kjaer, L.; Pigosso, D.C.A.; Hauschild, M.Z.; … Moreno, M., 2019. Evaluating the Environmental Performance of a Product/Service-System Business Model for Merino Wool Next-to-Skin Garments: The Case of Armadillo Merino®. Sustainability (11), 5854. https://doi.org/10.3390/su11205854

Berton, M., Bittante, G., Zendri, F., Ramanzin, M., Schiavon, S., Sturaro, E., 2020. Environmental impact and efficiency of use of resources of different mountain dairy farming systems. Agric. Syst., 181, 102806. https://doi.org/10.1016/j.agsy.2020.102806

Billen G, Lassaletta L, Garnier J. 2014. A biogeochemical view of the global agro-food system: Nitrogen flows associated with protein production, consumption and trade. Glob Food Sec 3(3): 209–219. https://doi.org/10.1016/j.gfs.2014.08.003

Borrello, M.; Pascucci, S.; Cembalo, L. Three Propositions to Unify Circular Economy Research: A Review. Sustainability 2020, 12, 4069

Boxmeer, E., Modernel, P., Viets, T., 2021. Environmental and economic performance of dutch dairy farms on peat soil. Agric. Syst. (193), 103243. https://doi.org/10.1016/j.agsy.2021.103243

Brankatschk, G. and Finkbeiner, M., 2014. Application of the Cereal Unit in a new allocation procedure for agricultural life cycle assessments. J. Clean. Prod. 73. 72–79. https://doi.org/10.1016/j.jclepro.2014.02.005.

Campos, I., Valente, LMP., Matos, E., Marques, P., Freire, F., 2019. Life-cycle assessment of animal feed ingredients: poultry fat, poultry byproduct meal and hydrolyzed feather meal, J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2019.119845

Castanheira, E. and Freire, F., 2013. Greenhouse gas assessment of soybean production: Implications of land use change and different cultivation systems. J. Clean. Prod. 49-60. https://doi.org/10.1016/j.jclepro.2013.05.026.

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. https://doi.org/10.1007/s11367-016-1201-y.

Colley, T., Valerian, J.,Hauschild, M., Olsen, S., Birkved, M., 2021. Addressing Nutrient Depletion in Tanzanian Sisal Fiber Production Using Life Cycle Assessment and Circular Economy Principles, with Bioenergy Co-Production. Sustainability. 13. https://doi.org/10.3390/su13168881.

Commoner, B. The Closing Circle. Nature, Man and Technology; Alfred, A., Ed.; Knopf, Inc.: New York, NY, USA, 1971

Cooper, W.W, Seiford, L.M, Tone, K., 2007. A Comprehensive Text with Models, Applications, References and DEA-Solver Software. Springer, New York. https://doi.org/10.1007/978-0-387-45283-8

Costantini, M., Lovarelli, D., Orsi, L., Ganzaroli, A., Ferrante, V., Febo, P., ... Bacenetti, J., 2020. Investigating on the environmental sustainability of organic animal products? The case of organic eggs. J. Clean. Prod. 274. 123046. https://doi.org/10.1016/j.jclepro.2020.123046.

Curran, M.A., 2008. Development of life cycle assessment methodology: a focus on co-product allocation.

De Vries, M. and de Boer, I.J.M., 2010. Comparing Environmental Impacts for Livestock Products: A Review of Life Cycle Assessments. Livest Sci., 128, 1-11. https://doi.org/10.1016/j.livsci.2009.11.007

Diaz, F., Vignati, J.A., Marchi, B., Paoli, R., Zanoni, S., Romagnoli, F., 2021. Effects of energy efficiency measures in the Beef cold chain: A life cycle-based study. Environ. Clim. Technol., 25:343-355. https://doi.org/10.2478/rtuect-2021-0025.

Ding, T., Bourrelly, S., Achten, W., 2021. Application of territorial emission factors with open-access data—a territorial LCA case study of land use for livestock production in Wallonia. Int. J. Life Cycle Assess. 26. 1-14. https://doi.org/10.1007/s11367-021-01949-3.

Dourmad, J.Y., Guilbaud, T.,Tichit, M., Bonaudo. T., 2019. Les productions animales dans la bioéconomie. INRA Productions Animales, Paris: INRA, 2019, 32 (2), pp.205-220. https://doi.org/10.20870/productions-animales.2019.32.2.2485

Duchin, F., Levine, S., 2011. Sectors may use multiple technologies simultaneously: the rectangular choice-of-technology model with binding factor constraints. Econ. Syst. Res. 23, 281–302. https://doi.org/10.1080/09535314.2011.571238.

Esposito, B., Sessa, M.R., Sica, D., Malandrino, O., 2020. Towards Circular Economy in the Agri-Food Sector. A Systematic Literature Review. Sustainability. 12(18):7401. https://doi.org/10.3390/su12187401.

Esteves, E.M.M., Brigagão, G.V., Cláudia R.V. Morgado, C.R.V., 2021. Multi-objective optimization of integrated crop-livestock system for biofuels production: A life-cycle approach, Renew. Sust. Energ. Rev., Volume 152, 111671, ISSN 1364-0321. https://doi.org/10.1016/j.rser.2021.111671

European Commission. Communication from the commission to the European parliament, the council, the European economic and social committee and the committee of the regions. In Closing the Loop—An EU Action Plan for the Circular Economy; European Commission: Brussels, Belgium, 2015

European Parliament. Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 Amending Directive 2008/98/EC on Waste. Available online: https://eur- ex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX: 32018L0851&from=EN

FAO. 2016. Environmental performance of large ruminant supply chains – Guidelines for assessment – Version 1. Livestock Environmental Assessment and Performance Partnership (FAO LEAP). Rome.

FAO. 2016. Greenhouse gas emissions and fossil energy use from small ruminant supply chains – Guidelines for assessment – Version 1. Livestock Environmental Assessment and Performance Partnership (FAO LEAP). Rome.

FAO, 2018. Nutrient flows and associated environmental impacts in livestock supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, FAO.

FAO. 2020. Environmental performance of feed additives in livestock supply chains – Guidelines for assessment – Version 1. Livestock Environmental Assessment and Performance Partnership (FAO LEAP). Rome.

FAO, IFAD, UNICEF, WFP and WHO. 2023. The State of Food Security and Nutrition in the World 2023. Urbanization, agrifood systems transformation and healthy diets across the rural–urban continuum. Rome, FAO. https://doi.org/10.4060/cc3017en

Finkbeiner, M., Ackermann, R., Bach, V., Berger, M., Brankatschk, G., Chang, Y., Grinberg, M., Lehmann, A., Martínez-Blanco, J., Minkov, N., Neugebauer, S., Scheumann, R., Schneider, L., Wolf, K., 2014. Challenges in Life Cycle Assessment: An Overview of Current Gaps and Research Needs. https://doi.org/10.1007/978-94-017-8697-3.

Florindo, T., Medeiros, G., Ruviaro, C., Pinto, A., 2019. Multicriteria Decision-Making And Probabilistic Weighing Applied To Sustainable Assessment Of Beef Life Cycle. J. Clean. Prod.. 242. 118362. https://doi.org/10.1016/j.jclepro.2019.118362.

Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., ... Tempio, G. 2013. Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities. Food and Agriculture Organization of the United Nations (FAO), Rome.

Ghisellini, P., Protano, G.,Viglia, S., Gaworski, M., Setti, M., Ulgiati, S., 2015. Integrated Agricultural and Dairy Production within a Circular Economy Framework. A Comparison of Italian and Polish Farming Systems. J. Environ. Account Ma. 2. 367-384. https://doi.org/10.5890/JEAM.2014.12.007.

Ghisellini, P., Setti, M. & Ulgiati, S., 2016. Energy and land use in worldwide agriculture: an application of life cycle energy and cluster analysis. Environ Dev Sustain 18, 799–837. https://doi.org/10.1007/s10668-015-9678-2.

Goglio, P., Knudsen, M.T., Van Mierlo, K., Röhrig, N., Fossey, M., Maresca, A., ... Smith, L.G., 2023. Defining common criteria for harmonizing life cycle assessments of livestock systems. CLPL 4, 100035. doi: 10.1016/j.clpl.2023.100035

Goglio, P., Williams, A., Balta-Ozkan, N., Harris, N.R.P., Williamson, P., Huisingh, D., ... Tavoni, M., 2019. Advances and challenges of Life Cycle Assessment (LCA) of Greenhouse Gas Removal Technologies to Fight Climate Changes. J. Clean. Prod. 118896 doi: 10.1016/j.jclepro.2019.118896.

Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2017. Addressing crop interactions within cropping systems in LCA. Int. J. Life Cycle Assess. 1–9 doi: 10.1007/s11367-017-1393-9.

González-García, S., Baucells, F., Feijoo, G., Moreira, M., 2016. Environmental performance of sorghum, barley and oat silage production for livestock feed using life cycle assessment. Resour. Conserv. Recycl. 111. 28-41. https://doi.org/10.1016/j.resconrec.2016.04.002.

Grassauer, F., Herndl, M., Nemecek, T., Fritz, C., Guggenberger, T., Steinwidder, A., Zollitsch, W., 2022. Assessing and improving eco-efficiency of multifunctional dairy farming: The need to address farms' diversity. J. Clean. Prod. 338. 130627. https://doi.org/10.1016/j.jclepro.2022.130627.

Herrero, M., Henderson, B., Havlík, P. et al. Greenhouse gas mitigation potentials in the livestock sector. Nature Clim Change 6, 452–461 (2016). https://doi.org/10.1038/nclimate2925

Hoekman, S., Broch, A., Liu, X., 2017. Environmental implications of higher ethanol production and use in the U.S.: A literature review. Part I – Impacts on water, soil, and air quality. Renew. Sust. Energ. Rev. 81. https://doi.org/10.1016/j.rser.2017.05.050.

Houssard, C., Maxime, D., Pouliot, Y., Margni, M., 2021. Allocation is not enough! A system boundaries expansion approach to account for production and consumption synergies: The environmental footprint of Greek yogurt. J. Clean. Prod. (283). https://doi.org/10.1016/j.jclepro.2020.124607.

Huysveld S, Schaubroeck T, De Meester S, Sorgeloos P, Van Langenhove H, Van linden V, Dewulf J, 2013. Resource use analysis of Pangasius aquaculture in the Mekong Delta in Vietnam using Exergetic Life Cycle Assessment, J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2013.01.024.

ISO, 2015.S-EN ISO 14001 Environmental Management Systems – Requirements with guidance for use. International Organization for Standardization, Geneva.

ISO, 2006a.SS-EN ISO 14040 Environmental Management- Life Cycle Assessment, Principles and Framework. International Organization for Standardization, Geneva.

ISO, 2006b.SS-EN ISO 14044 Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organization for Standardization, Geneva.

Jones, D., Vlieg, M., Ashar, S., Friend, L., Gomez, C.C. (2022). Learning to quantify positive futures. International Journal of Environmental Impacts, Vol. 5, No. 2, pp. 128-145. https://doi.org/10.2495/EI-V5-N2-128-145

JRC, 2010. International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition. (No. EUR 24708 EN). European Commission, Joint Research Centre, Publications Office of the European Union; 2010, Luxembourg.

Laso, J., Margallo, M., Garcia Herrero, I., Fullana-i-Palmer, P., Bala, A., Gazulla, C., … Aldaco, R., 2018. Combined application of Life Cycle Assessment and linear programming to evaluate food waste-to-food strategies: Seeking for answers in the nexus approach. Waste Manage. 80. 186-197. https://doi.org/10.1016/j.wasman.2018.09.009.

Laurenti, R., Moberg, A., Stenmarck, A., 2017. Calculating the pre-consumer waste footprint: A screening study of 10 selected products. Waste Manag. Res., 35:65-78. https://doi.org/ 10.1177/0734242X16675686.

Leip A, Ledgard S, Uwizeye A, Palhares JCP, Aller MF, Amon B, … Wang Y., 2019. The value of manure - Manure as co-product in life cycle assessment. J Environ Manag. Jul 1;241:293-304. https://doi.org/10.1016/j.jenvman.2019.03.059.

Linder, M, Sarasini, S, van Loon, P, 2017. A Metric for Quantifying Product-Level Circularity: Product-Level Circularity Metric. J. of Ind. Ecol. 21, 545–558. https://doi.org/10.1111/jiec.12552

Macombe, C., Loeillet, D., Gillet, C., 2018. Extended community of peers and robustness of social LCA. Int. J. Life Cycle Assess. 23, 492–506. https://doi.org/10.1007/s11367-016-1226-2.

Maiolo, S., Parisi, G., Biondi, N., Lunelli, F., Tibaldi, E., Pastres, R., 2020. Fishmeal partial substitution within aquafeed formulations: life cycle assessment of four alternative protein sources. Int. J. Life Cycle Assess. 17. https://doi.org/10.1007/s11367-020-01759-z.

Marton, S., Zimmermann, A., Kreuzer, M., Gaillard, G., 2016. Comparing the environmental performance of mixed and specialised dairy farms: The role of the system level analysed. J. Clean. Prod. 124. https://doi.org/10.1016/j.jclepro.2016.02.074.

McAuliffe, G., Chapman, D., Sage, C., 2016. A thematic review of life cycle assessment (LCA) applied to pig production. Environ. Impact Assess. Rev. 56. 12-22. https://doi.org/ 10.1016/j.eiar.2015.08.008.

McLaren, S., Berardy, A., Henderson, A., Holden, N., Huppertz, T., Jolliet, O., … van Zanten, H. 2021. Integration of environment and nutrition in life cycle assessment of food items: opportunities and challenges. Rome, FAO.

Moraine, M., Duru, M., & Therond, O. (2017). A social-ecological framework for analyzing and designing integrated crop–livestock systems from farm to territory levels. Renew. Agric. and Food Syst., 32(1), 43-56. https://doi.org/10.1017/S1742170515000526.

Mosnier, C., Jarousse, A., Madrange, P., Balouzat, J., Guillier, M., Mertens, A., ... Veysset, P., 2021. Evaluation of the contribution of 16 European beef production systems to food security. Agric. Syst., Elsevier Masson, 2021, 190, pp.103088. https://doi.org/10.1016/j.agsy.2021.103088.

Mottet, A., de Haan, C., Falcucci, A., Tempio, G., Opio, C., Gerber, P., 2017. Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. Glob. Food Sec. 14, 1–8. https://doi.org/10.1016/j.gfs.2017.01.001.

Mourjane I. et Fosse J. (2021), « La biomasse agricole : quelles ressources pour quel potentiel ? », Note de synthèse, n° 2021-03, juillet. »

Mullender, S.M., Sandor, M., Pisanelli, A., Kozyra, J., Borek, R., Ghaley, B.B., ... Smith, L.G., 2020. A delphi-style approach for developing an integrated food/non-food system sustainability assessment tool. Environ. Impact Assess. Rev. 84, 106415. https://doi.org/ 10.1016/j.eiar.2020.106415.

Muscat, A., de Olde, E.M., de Boer, I.J.M., Ripoll-Bosch, R., 2020. The battle for biomass: A systematic review of food-feed-fuel competition. Glob. Food Sec. 25, 100330. https://doi. org/10.1016/j.gfs.2019.100330.

Navarro, D., Wu, J., Lin, W., Fullana-i-Palmer, P., Puig, R., 2020. Life cycle assessment and leather production. Journal of Leather Science and Engineering. 2. https://doi.org/10.1186/s42825-020-00035-y.

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

Nikodinovic-Runic, J., Guzik, M., Kenny, S.T., Babu, R., Werker, A., O Connor, K.E., Chapter Four - Carbon-Rich Wastes as Feedstocks for Biodegradable Polymer (Polyhydroxyalkanoate) Production Using Bacteria, Editor(s): Sima Sariaslani, Geoffrey M. Gadd, Advances in Applied Microbiology, Academic Press, Volume 84, 2013, Pages 139-200, https://doi.org/10.1016/B978-0-12-407673-0.00004-7.

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, 399–409. https://doi.org/10.1016/ j.jclepro.2016.06.071.

Obersteiner, M., Walsh, B., Frank, S., Havlik, P., Cantele, M., Liu, J., ... van Vuuren, D., 2016. Assessing the land resource-food price nexus of the Sustainable Development Goals. Sci. Adv. 2, e1501499–e1501499. https://doi.org/10.1126/sciadv.1501499

Ogino, A.,Thu, N., Hosen, Y., Izumi, T., Suzuki, T., Sakai, T., ... Kawashima, T., 2021. Environmental impacts of a rice-beef-biogas integrated system in the Mekong Delta, Vietnam evaluated by life cycle assessment. J. Environ. Manage. 294. 112900. https://doi.org/10.1016/j.jenvman.2021.112900.

Oldfield, T.L., White, E., Holden, N.M., 2016. An environmental analysis of options for utilising wasted food and food residue, J. Environ. Manage, Volume 183, Part 3, 2016, Pages 826-835, ISSN 0301-4797, https://doi.org/10.1016/j.jenvman.2016.09.035.

Oliveira, M., Cocozza, A., Zucaro, A., Santagata, R., Ulgiati, S., 2021. Circular economy in the agro-industry: Integrated environmental assessment of dairy products. Renew. Sust. Energ. Rev. (148). https://doi.org/10.1016/j.rser.2021.111314

Olofsson, J., Börjesson, P., 2018. Residual biomass as resource - Life-cycle environmental impact of wastes in circular resource systems. J. Clean. Prod. (196), 997-1006. https://doi.org/10.1016/j.jclepro.2018.06.115

Oosting, S., van der Lee, J., Verdegem, M. et al. Farmed animal production in tropical circular food systems. Food Sec. 14, 273–292 (2022). https://doi.org/10.1007/s12571-021-01205-4

Oteros-Rozas, E., Martín-López, B., González, J.A., Plieninger, T., López, C.A, & Montes, C. 2013a. Socio-cultural valuation of ecosystem services in a transhumance social-ecological network. Reg. Environ. Change 14(4): 1269-1289.

Stanley, P., & Rowntree, J., 2018, Impacts of soil carbon sequestration on life cycle greenhouse gas emissions in Midwestern USA beef finishing systems, Agric. Syst., Volume 162. https://doi.org/10.1016/j.agsy.2018.02.003.

Paolotti, L., Boggia, A., Castellini, C., Rocchi, L., Rosati, A., 2016. Combining livestock and tree crops to improve sustainability in agriculture: a case study using the LCA approach. J. Clean. Prod. 131. https://doi.org/10.1016/j.jclepro.2016.05.024.

Paramesh, V., Ranjan, P., Eaknath, C., Sreekanth, G.B., Chethan, K., Gokuldas, P.P., ... Natesan, R., 2019. Sustainability, energy budgeting, and life cycle assessment of crop- dairy-fish-poultry mixed farming system for coastal lowlands under humid tropic condition of India. Energy. 116101. https://doi.org/10.1016/j.energy.2019.116101.

BSI, 2011. PAS 2050. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution, London.

Payraudeau, S., van der Werf, H.M.G., 2005. Environmental impact assessment for a farming region: a review of methods. Agric. Ecosyst. Environ. 107, 1–19. URL:833. https://doi.org/10.1016/j.agee.2004.12.012

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 (1), 74e86. https://doi.org/10.1007/s11367-014-0812-4

Petit, G., Sablayrolles, C., Yannou-Le Bris, G. Combining eco-social and environmental indicators to assess the sustainability performance of a food value chain: A case study. J. Clean. Prod. 2018, 191, 135–143. https://doi.org/10.1016/j.jclepro.2018.04.156

Place, Sara E. PhD; Myrdal Miller, Amy MS, RDN, FAND. Beef Production: What Are the Human and Environmental Impacts?. Nutr. Today: 9/10 2020 - Volume 55 - Issue 5 - p 227-233. https://doi.org/ 10.1097/NT.0000000000000432

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science (80-.). 360, 987–992. https://doi.org/10.1126/science.aaq0216.

Reckmann, K., Blank, R., Traulsen, I., Krieter, J., 2016. Comparative life cycle assessment (LCA) of pork using different protein sources in pig feed. Archiv fur Tierzucht. 59. 27-36. 10.5194/aab-59-27-2016. https://doi.org/10.5194/aab-59-27-2016.

Roelcke, M., Heimann, L., Hou, Y., Guo, J., Xue, Q., JIA, W., ... Zhang, F., 2019. Phosphorus status, use and recycling in a Chinese peri-urban region with intensive animal husbandry and cropping systems. Front. Agric. Sci. Eng. 6. 10.15302/J-FASE-2019286. https://doi.org/10.15302/J-FASE-2019286.

Rosner, B. (2011). Fundamentals of Biostatistics, 7th Edition. Australia, Brazil, Japan, Mexico Korea, Singapore, UK, USA: Brooks/Cole Cengage Learning.

Sadhukhan, J., Dugmore, T.I.J., Matharu, A., Martinez-Hernandez, E., Aburto, J., Rahman, P.K.S.M., Lynch, J., 2020. Perspectives on "Game Changer" Global Challenges for Sustainable 21st Century: Plant-Based Diet, Unavoidable Food Waste Biorefining, and Circular Economy. Sustainability (12), 1976. https://doi:10.3390/su12051976.

Schaubroeck, T., Schaubroeck, S., Heijungs, R., Zamagni, A., Brandão, M., & Benetto, E., 2021. Attributional & Consequential Life Cycle Assessment: Definitions, Conceptual Characteristics and Modelling Restrictions. Sustainability, 13(13).

Schestak, I., Styles, D., Black, K., Williams, A.P., 2022. Circular use of feed by-products from alcohol production mitigates water scarcity. Sustain. Prod. Consum, 30:158-170. https://doi.org/10.1016/j.spc.2021.11.034.

Sekaran, U., Lai, L., Ussiri, D.A.N., Kumar, S., Clay, S., 2021. Role of integrated crop-livestock systems in improving agriculture production and addressing food security – A review, J. Agric. Food Res., Volume 5. https://doi.org/10.1016/j.jafr.2021.100190.

Sneessens, I., Veysset, P., Benoit, M., Lamadon, A., Brunschwig, G., 2016. Direct and indirect impacts of crop-livestock organization on mixed crop-livestock systems sustainability: A model-based study. Animal. -1. 1-12. https://doi.org/10.1017/S1751731116000720.

Siddiqui, Z., Hagare, D., Jayasena, V., Swick, R., Rahman, M.M., Boyle, N., Ghodrat, M., 2021. Recycling of food waste to produce chicken feed and liquid fertiliser. Waste Manage., 131/386-393. https://doi.org/10.1016/j.wasman.2021.06.016.

Springer, N., Schmitt, J., 2018. The price of byproducts:Distinguishing co-products from waste using the rectangular choice-of-technologies model. Resour. Conserv. Recycl., 138:231-237. https://doi.org/10.1016/j.resconrec.2018.07.034

Stanchev, P., Vasilaki, V., Egas, D., Colon, J., Ponsá, S., Katsou, E., 2020. Multilevel environmental assessment of the anaerobic treatment of dairy processing effluents in the context of circular economy, J. Clean. Prod. (261), ISSN 0959-6526, https://doi.org/10.1016/j.jclepro.2020.121139

Steinfeld, Henning & Gerber, Pierre J. & Wassenaar, Tom & Castel, Vincent & Rosales, Mauricio & De haan, Cornelis. (2006). Livestock's Long Shadow: Environmental Issues and Options. United Nations Food and Agriculture Organization. ISBN: 978-92-5-105571-7

Stillitano, T.; Spada, E.; Iofrida, N.; Falcone, G.; De Luca, A.I., 2021. Sustainable Agri-Food Processes and Circular Economy Pathways in a Life Cycle Perspective: State of the Art of Applicative Research. Sustainability, 13, 2472. https://doi.org/10.3390/su13052472

Talwar, N., Holden, N.M. The limitations of bioeconomy LCA studies for understanding the transition to sustainable bioeconomy. Int J Life Cycle Assess 27, 680–703 (2022). https://doi.org/10.1007/s11367-022-02053-w

TEEB 2010. The Economics of Ecosystems and Biodiversity. Ecological and Economic Foundations. Kumar, P (ed). Earthscan, London and Washington

Teston, M., Villalba, D., Berton, M., Ramanzin, M., Sturaro, E., 2020. Relationships between organic beef production and agro-ecosystems in mountain areas: The case of Catalan Pyrenees. Sustainability, 12, 9274. https://doi.org/10.3390/su12219274.

Thévenot, A., Aubin, J., Tillard, E.,Vayssières, J., 2013. Accounting for farm diversity in Life Cycle Assessment studies - The case of poultry production in a tropical island. J. Clean. Prod. 57. 280-292. https://doi.org/110.1016/j.jclepro.2013.05.027.

van Hal, O., Weijenberg, A., Boer, I.J.M., Zanten, H., 2019. Accounting for feed-food competition in environmental impact assessment: Towards a resource efficient food-system. J. Clean. Prod. 240. 118241. https://doi.org/ 10.1016/j.jclepro.2019.118241

Verduna, T., Blanc, S., Merlino, V.M., Cornale, P., Battaglini, L.M., 2020. Sustainability of four dairy farming scenarios in an alpine environment: The case study of toma di Lanzo cheese. Front. Vet. Sci., 7, 569167. . https://doi.org/10.3389/fvets.2020.569167.

Vogel, E., Martinelli, G., Dalzotto Artuzo, F., 2021. Environmental and economic performance of paddy field-based crop-livestock systems in Southern Brazil. Agric. Syst.. 190. 103109. https://doi.org/10.1016/j.agsy.2021.103109

Ward, S., White, E., Holden, N., Oldfield, T., 2016. The 'circular economy' applied to the agriculture (livestock production) sector – discussion paper. Available from https://ec.europa.eu/information_society/newsroom/image/document/2016-48/ward_-_circular_economy_applied_to_the_livestock_production_sector___brussels__2_40231.pdf (last accessed 08 September 2023)

Weiler, V., Udo, H.M.J., Viets, T.C., Crane, T., Boer, I.J.M., 2014. Handling multi-functionality of livestock in a life cycle assessment: The case of smallholder dairying in Kenya. Current Opinion in Environmental Sustainability. 8. 29-38. https://doi.org/10.1016/j.cosust.2014.07.009.

Wiedmann, T., Wilting, H., Lutter, S., Palm, V., Giljum, S., Wadeskog, A., & Nijdam, D., 2009. Development of a methodology for the assessment of global environmental impacts of traded goods and services (SKEP ERA-NET Project EIPOT) [Technical report]. Stockholm, Sweden; Bilthoven, Netherlands; Wien, Austria: Stockholm Environment Institute (SEI), Netherlands Environmental Assessment Agency (PBL), Sustainable Europe Research Institute (SERI), Statistics Sweden, Environmental Accounting Unit.

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.

Wu, S., Liu, X., Wang, L., Chen, J., Zhou, P., Shao, C., (2022. Integrating life cycle assessment into landscape studies: a postcard from Hulunbuir. Landsc. Ecol. 37. https://doi.org/10.1007/s10980-021-01396-3.

Yan Z, Li W, Yan T, Chang S, Hou F. 2019. Evaluation of energy balances and greenhouse gas emissions from different agricultural production systems in Minqin Oasis, China. PeerJ., 7:e6890 https://doi.org/10.7717/peerj.6890.

Yue, Q., Guo, P., Wu, H., Wang, Y., Zhang, C., 2021. Towards sustainable circular agriculture: An integrated optimization framework for crop-livestock-biogas-crop recycling system management under uncertainty. Agric. Syst., 196, 103347. https://doi.org/10.1016/j.agsy.2021.103347.

Zampori, L., & Pant, R. (2019). Suggestions for updating the Product Environmental Footprint (PEF) method. Luxembourg: EUR 29682 EN, Publications Office of the European Union.

Zanten, H., Bikker, P., Meerburg, B., Boer, I.J.M., 2018. Attributional versus consequential life cycle assessment and feed optimization: alternative protein sources in pig diets. Int. J. Life Cycle Assess. 23. https://doi.org/10.1007/s11367-017-1299-6.

Zanten, H., Müller, A., Frehner, A., 2022. Land use modeling: from farm to food systems. https://doi.org/10.1016/B978-0-12-822112-9.00011-4.

Zhang, X., Ma, F., 2012. Application of Life Cycle Assessment in Agricultural Circular Economy. App. Mech. Mater. 260-261. 1086-1091. https://doi.org/10.4028/www.scientific.net/AMM.260-261.1086.

SG2: Biodiversity
Toward better biodiversity impact assessment of agricultural land management through life cycle assessment: a systematic review

Huayang Zhen [a,*], Pietro Goglio [b], Fatemeh Hashemi [a], Christel Cederberg [c], Maxime Fossey [d], Marie Trydeman Knudsen [a]

[a] Department of Agroecology, Aarhus University, Blichers Allè 20, 8830 Tjele, Denmark;

[b] Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy;

[c] Division Physical Resource Theory, Chalmers University of Technology, 41296 Gothenburg, Sweden;

[d] Institut de l'élevage (IDELE), 149 rue de Bercy, 75012 Paris, France.

*Corresponding author: Huayang Zhen (huayang.zhen@agro.au.dk)

SG2 report is in preparation for the journal Environmental Science and Technology.

Abstract:

Agricultural intensification has driven global biodiversity loss through land management change. However, there is no consensus on assessing the biodiversity impacts of land management change using life cycle assessment (LCA). This study conducts a systematic literature review of LCA methods of assessing biodiversity impacts to evaluate their quality and identify research needs for incorporating land management change in LCA. We evaluated 7 expert scoring-based (ESB) and 19 biodiversity indicator-based (BIB) methods that assessed biodiversity impacts of land management change. Generally, BIB methods outperformed ESB methods in general criteria, especially in robustness (95% higher). The available methods focus on various biodiversity levels and aspects, but each considers limited biodiversity characteristics and cannot represent the comprehensive biodiversity concept. BIB methods tend to use land management intensity levels, while ESB methods focus on specific land management practices. Despite their advantages, neither approach is sufficient for fully capturing biodiversity impacts across supply chains. For future studies, it is advisable to (1) model the direct (on-farm) impacts of land management change at the midpoint level; (2) establish cause-effect relationships between crucial land management practices and biodiversity indicators and distinguish between direct (on-site) and indirect (off-site) biodiversity impacts resulting from land management change; (3) characterise land-use intensity levels based on specific land management practices and include the positive impact from agroecological practices. This review highlights the current state of LCA methods and suggests improvements to better account for the complexity of biodiversity impacts from agricultural land management.

Keywords:

biodiversity, life cycle assessment, land management change, agroecosystem, expert opinion

Introduction

Addressing the decline of biodiversity is urgently needed. Despite the crucial role biodiversity plays in ecosystem functioning and human well-being[1], current extinction rates are staggering, roughly 1000 times that of the likely background ones[2]. Approximately 25% of the remaining species are threatened by various direct drivers of biodiversity loss, such as land/sea use change, direct exploitation, climate change, pollution, etc.[3] Agroecosystems, including cropland and pastures, cover 46% of the Earth's land surface[4] and pose risks to 62% of globally threatened species[5], playing a vital role in global biodiversity conservation efforts[6]. Agricultural land is predicted to expand or undergo intensification to feed the growing world population and meet the increasing per-capita consumption[7], which will have significant biodiversity impacts due to land-use class change and land management change[8]. At the same time, agriculture is also dependent on biodiversity, e.g., pollination, natural enemies of pests, and turnover of organic matter in the soil for C and N cycles[9]. Even though agricultural land use threatens biodiversity, the degree to which it harms local biodiversity varies (e.g., grazed meadows versus intensive wheat production). Since the agroecosystem covers a large proportion of the Earth's surface, the biodiversity losses due to agricultural management are crucial[10]. Besides, food consumption accounts for the largest share (40%) of global biodiversity loss among human consumption activities[11]. Understanding and mitigating these losses is vital for the sustainable development of agriculture. Properly assessing the biodiversity impacts of agricultural products is essential to achieving the UN Sustainable Development Goal of Responsible Production and Consumption (SDG 12).

Life cycle assessment (LCA) is a method used to evaluate the environmental impacts of products/services. LCA was initially developed to assess the impacts of extractions and emissions associated with material and energy flows of industrial products[12]. After the UNEP Life Cycle Initiative issued a framework on global land use impact assessment[13], many research studies have applied it to assess biodiversity impacts, e.g., Species-Area Relationship (SAR) method[14, 15] and Countryside SAR (CSAR) method[16], etc. Through methodological development, many operational LCA methods, such as ReCiPe 2016, LC-impact, Impact world+, Stepwise, and Ecoscarcity 2013, can already capture biodiversity impacts[17]. These methods introduce biodiversity as an endpoint impact category (except Ecoscarcity 2013), which means biodiversity is influenced by climate change, land use change, pollution, etc. However, the currently operational LCA methods seldom evaluate the direct biodiversity impacts of land management change[17, 18].

Land management characterises variation in how land is used 'within' the different land-use classes[7, 8, 19]. In the agricultural sector, land management practices include tillage, seeding, weeding, applying fertilisers or pesticides, irrigating, harvesting, mowing, grazing livestock on land with different livestock loads, etc.[20] Agricultural management practices can have direct and indirect impacts on biodiversity, either positive or negative[21]. Agroecological practices can increase the biodiversity compared to the conventional agricultural practices. For example, moderate and light grazing enhance general biodiversity, while heavy grazing has the opposite results[22]. However, LCA methods mainly focus on the negative impacts of biodiversity instead of incorporating the positive effects of land management[17]. Ignoring agricultural management practices, especially agroecological practices that positively impact biodiversity, can bias the results of biodiversity impact assessment of food products, which might mislead the production and consumption. Integrating land management into biodiversity models would help to achieve more precise prediction and evaluation. At the World Biodiversity Forum 2020 in Davos, LCA researchers agreed that land management needs to be considered to better understand biodiversity impacts[23].

Curran, et al. [24] evaluated the LCA methods of biodiversity with defined criteria, encompassing model completeness, biodiversity representation, etc., and recommended seven best practices that can be implemented immediately to improve the current LCA methodologies on biodiversity. Crenna, et al. [17] reviewed the operational and non-operational LCA methodologies that can assess the biodiversity impacts of value chains and found that the current methodologies are poor at capturing the complexities of biodiversity. More recently, Damiani, et al. [25] evaluated the most recent LCA methodologies on biodiversity and suggested some research perspectives to overcome the current research gap in this area, e.g., including more biodiversity loss drivers, increasing ecosystem and taxonomic coverage, etc. Gabel, et al. [18] specifically focused on the challenges of including biodiversity impacts with LCA in the agricultural sector, focusing on the functional unit, biodiversity aspects and indicators, reference condition, global applicability, and differentiating between agricultural intensities. Despite the importance of including the biodiversity impacts of agricultural management practices in LCA, previous review papers scarcely focused on this topic. Therefore, it is essential to review the state of the art and research needs pertaining to assessing biodiversity impacts resulting from land management practices.

Regarding data origin, there are two types of LCA methodologies to assess the biodiversity impacts: expert scoring-based (ESB) and biodiversity indicator-based (BIB) methods. The ESB covered methods that evaluate biodiversity impacts with experts' judgments (e.g., AgBalance[26], Swiss Agricultural LCA—Biodiversity[27, 28], etc.); in this group we included also SALCA-Biodiversity even it was based on expert judgement and literature. Instead the BIB method encompassed methods that generally reflect the biodiversity impacts with direct algorithms for biodiversity indicators and/or biodiversity models (e.g., Species area relationship[14, 15]). Expert judgment can be easily applied to any system when expert opinion is available[29], especially for establishing the cause-effect relationship between land management practices and the impacts of biodiversity. However, the quality of the two types of LCA methods in assessing biodiversity impacts and their potential for integrating land management practices are yet to be evaluated.

This systematic review aims to investigate how the LCA methods in the agricultural sector evaluate the impacts of land management change on biodiversity. Specifically, this paper will 1) evaluate and compare the quality of the ESB and BIB methods with a set of criteria and highlight their strengths and weaknesses; 2) explore the state of the art of incorporating land management change in LCA methods of biodiversity impacts; and 3) identify research needs for incorporating the land management change into LCA methods for assessing the biodiversity impacts of agroecosystems. This paper will provide insights into incorporating agricultural management practices into LCA methods of biodiversity impact assessment.

Materials and methods

We conducted a systematic review of the methodologies of biodiversity impact assessment with LCA. To review relevant LCA studies, a selection of publications available online no later than March 3rd 2022, was made via Web of Science, Scopus, and Google Scholar with the keywords ("LCA" OR "Life cycle assessment" OR "life cycle analysis") AND ("biodiversity" OR "biodiversity impact assessment" OR "diversity" OR "biodiversity loss") AND ("livestock" OR "agricultur*" OR "farm*" or "land use" OR "land management" OR "management intensity" OR "habitat change" OR "habitat" OR "habitat loss" OR "species richness" OR "Species abundance" OR "fauna" OR "flora" OR "endangered species" OR "vulnerable species").

The setup of search criteria yielded 476 publications based on published scientific studies. After screening out the duplicate, non-English, non-LCA study, review, discussion, and fishery-related items, 111 publications remain (Fig.1). Following PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses) guidelines[30], the eligible studies 1) are original LCA methodology publications for evaluating biodiversity impacts, 2) include agricultural production or agricultural land use types, and 3) include the biodiversity loss drivers of land management change.

This systematic review focuses on land management change. However, a grey area exists between land-use class change and land management change, particularly when land management change is characterised by land management intensity levels (LMIL) (e.g., organic vs. conventional land, intensive vs. less intensive) rather than specific land management practices (LMP) (e.g., fertilisation, weeding, etc.). Land-use classes refer to a categorical distinction between different types of land use[31]. Land management change characterises variation in how land is used 'within' the different land-use classes[7, 8, 19]. To clarify, this paper defines a shift of LMIL as a land management change rather than a land use class change. Besides, this paper distinguishes cropland and grassland as two land use types according to the IPCC[32]. So, if a study focuses only on the two broad categories, it is considered land use class change. However, if it examines more specific land use types within cropland or grassland, such as irrigated and non-irrigated land, rice fields and other arable land, low and high grazing intensity, etc., it is considered land management change.

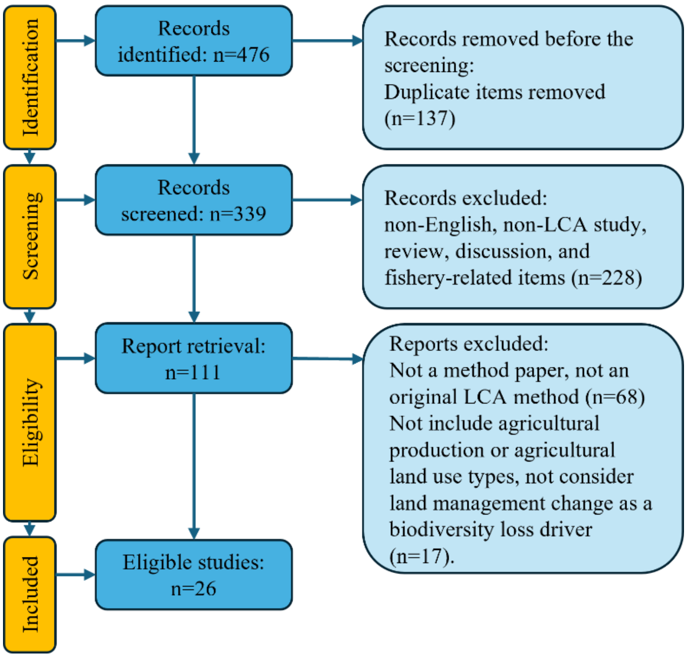


Fig.1 Literature screening and inclusion flowchart

This yielded a master bibliography of 26 studies (Table S1), which were classified into expert scoring-based (ESB) and biodiversity indicator-based (BIB) methods. Then, the methods were classified into subcategories based on the biodiversity models or methodological concepts applied. The ESB covered 7 methods under 4 method subcategories, while the BIB covered 19 methods under 9 method subcategories. Methods that applied SAR or CSAR but addressed different land use types or considered different species were considered distinct.

The methods were evaluated using general and specific criteria (Table 1). The criteria were developed by Goglio, et al. [33] using a modified DELPHI method, including several workshops among 21 LCA experts and two anonymous surveys. General criteria are employed to assess the quality of the reviewed method as an LCA methodology. Specific criteria are used to evaluate the quality of the reviewed method for assessing the biodiversity impacts. However, the specific criteria were slightly revised to better evaluate the reviewed methods. The changes are shown in italic text in Table S2. There are three or four levels for each criterion, where levels 1, 2, 3, and 4 correspond to scores 1, 2, 3, and 4, respectively (Table S2). Higher scores represent better performance. The first author evaluated and scored the reviewed methods with the predefined general and specific criteria. The scoring results of each method are shown in Table S1.

Table 1 A brief description of the general and specific criteria for evaluating the methods

Criteria type	Criteria	Brief description
General criteria	Transparency and reproducibility	Ability to allow reviewers to verify/review all data, calculations, and assumptions
	Completeness	Inclusion of material/energy flows and other environmental interventions, the data requirements, and the impact assessment methods
	Fairness and acceptance	Providing a level playing field across competing products, processes, and industries
	Robustness	Sensitivity, data quality, reliability, consistency, comparability, etc.
	Applicability	The ability to be used by a wide range of LCA practitioners
Specific criteria	*Predictability*	Predicting changes in biodiversity due to land management
	Inclusion of vulnerability and irreplaceability	Consideration of vulnerability and irreplaceability
	Inclusion of functional biodiversity	Characterisation of the functional biodiversity
	Species richness and diversity-accuracy	Data quality of the species richness and diversity
	Species richness and diversity - comprehensiveness	Capacity to capture the diversity and richness of all types of taxa and species
	Landscape continuity- accuracy	Capacity to capture the degree of landscape heterogeneity and connectivity

Note: Sourced and adjusted from Goglio, et al. [33]. Italicised criteria indicate changes from the original source.

To better describe the differences between the ESB and BIB, we extracted some crucial indicators of the LCA methods for biodiversity impact assessment from each method, including biodiversity level, biodiversity indicator, biodiversity-related data source, reference state, and taxon. Then, the frequency of each extracted indicator was calculated. Besides, the biodiversity representation of each method was evaluated by checking if the method considered the key biodiversity characteristics at the ecosystem (configuration, fragmentation, vulnerability, and irreplaceability) and species (affinity/sensitivity, functional diversity, vulnerability, and irreplaceability) levels.

Results and discussion

METHOD QUALITY EVALUATION

BIB methods performed better than ESB methods in terms of general criteria (Fig.2a). Compared to ESB methods, BIB exhibited a remarkably higher score in robustness (95%) because they are generally based on sophisticated biodiversity models. Furthermore, BIB methods showed slightly higher scores than ESB methods in completeness, applicability, and transparency and reproducibility. However, BIB is scored marginally lower than ESB in fairness and acceptance. Hence, BIB methods exhibited higher quality than LCA methods compared to ESB methods. Concerning specific criteria, BIB methods showed lower strengths than ESB methods in Predictability (-35%) and Inclusion of vulnerability and irreplaceability (-21%) (Fig.2b). Thus, BIB methods showed a lower ability to capture the land management practices and to reflect the biodiversity importance. BIB methods also scored lower than ESB methods in Landscape continuity-Accuracy (-18%) and Species richness and diversity-Comprehensiveness (-12%), indicating BIB methods have lower strength in modelling the impacts of land fragmentation and covering a broader range of species. Oppositely, BIB methods showed a considerably higher score in species richness and diversity-accuracy (100%) than ESB because their results are based on biodiversity models. Both types of methods showed few differences regarding the inclusion of functional biodiversity. Despite their lower accuracy and robustness, ESB methods can include more species and evaluate land management more flexibly than BIB methods.

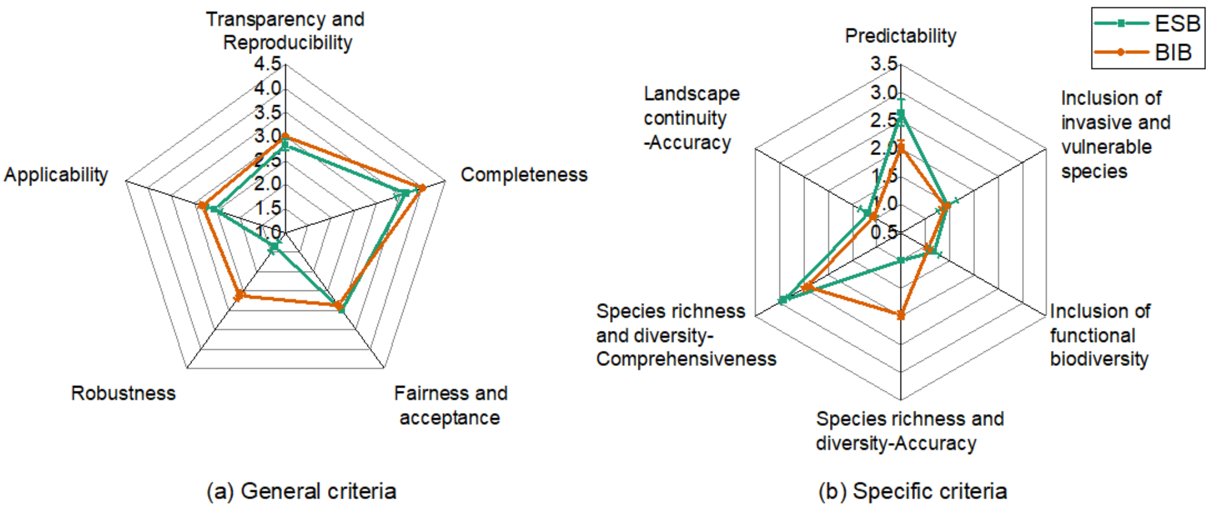


Fig. 2 Evaluation results of expert scoring (ESB, N=7) and biodiversity indicator-based (BIB, N=19) methods with general (a) and specific (b) criteria.

METHOD ATTRIBUTES

The reviewed methods covered three levels of biodiversity but mainly focused on the species level (81%) (Table 2). Notably, ESB methods covered more biodiversity levels than BIB methods. Although there is no universal agreement on biodiversity indicators, the most used measure of biodiversity is species richness (58%), the number of species at a given location and period[34]. Despite several limitations in using species richness as an indicator, it might be the most suitable indicator for biodiversity assessments due to its higher data availability and the lower data requirements than species abundance, etc.[35, 36]. All the ESB methods evaluated the biodiversity impacts with dimensionless scores instead of a biodiversity indicator.

All the ESB methods applied expert scoring data, while BIB methods mainly used data from databases (79%) and literature (53%). Expert scoring is expensive and time-consuming[37]. Reference states or values of biodiversity impact varied among the methods. Approximately 42% of the methods used semi- or natural or undisturbed ecosystems as a reference state, the type of land without or with little human perturbation. If the biodiversity of a land use type is lower than the reference state, there is damage to the ecosystem quality. Approximately 53% of the BIB methods used semi- or natural or undisturbed ecosystems as a reference. However, around 57% (4) of the ESB methods did not apply a reference state or value. Agricultural land occupation is necessary for food security and human well-being. It can be questioned whether the biodiversity of farmland should be compared to natural vegetation (e.g., natural forest) if the farm has existed for more than thousands of years[38]. Therefore, it is essential and important to establish reference states that are consistent with biodiversity targets that align with society's conservation frameworks[39].

Generally, the most evaluated taxon are Birds (46%), Mammals (42%), Vascular plants (31%), Amphibians (31%), and Reptiles (31%) (Table 2). Three of the 7 ESB methods did not specify the evaluated taxon but assessed the influenced organisms in general, similar to the two BIB methods, including Satoyama Index (SI)[40] and Spectral heterogeneity (SH)[41]. Most studies focused on terrestrial organisms, while few focused on aquatic species (especially ocean creatures), soil fauna, soil microbes, etc. The potential to generalise results from one well-studied species group to biodiversity is questionable[42]. ESB methods can include more species than BIB methods through experts' opinions or by assessing biodiversity impacts on general organisms.

Table 2 Frequency of the critical indicators applied in expert scoring based (ESB), biodiversity indicator-based (BIB), and all methods reviewed.

Indicators	Indicator values	ESB (N=7)	BIB (N=19)	Total (N=26)
Biodiversity level	Species	4 (57%) [a]	17 (89%)	21(81%)
	Ecosystem	2 (29%)	2 (11%)	4 (15%)
	Gene	2 (29%)	0 (0%)	2 (8%)
	Not applicable [b]	3 (43%)	0 (0%)	3 (12%)
Biodiversity indicator	Species richness	0 (0%)	15 (79%)	15(58%)
	Not applicable	7 (100%)	0 (0%)	7 (27%)
	Species abundance	0 (0%)	3 (16%)	3 (12%)
	Ecosystem diversity	0 (0%)	2 (11%)	2 (8%)
	Functional diversity	0 (0%)	1 (5%)	1 (4%)
	Functional evenness	0 (0%)	1 (5%)	1 (4%)
	Functional richness	0 (0%)	1 (5%)	1 (4%)
	the others	0	5	5
Biodiversity-related data sources	Database	1 (14%)	15 (79%)	16 (62%)
	Literature	0 (0%)	10 (53%)	10 (38%)
	Expert scoring	7 (100%)	0 (0%)	7 (27%)
	Direct observation	0 (0%)	2 (11%)	2 (8%)
Reference state	Semi- or natural or undisturbed ecosystem	1 (14%)	10 (53%)	11 (42%)
	Not applicable	4 (57%)	1 (5%)	5 (19%)
	Previous situation	0 (0%)	2 (11%)	2 (8%)
	Current situation	0 (0%)	2 (11%)	2 (8%)
	Regional average species richness	0 (0%)	2 (11%)	2 (8%)
	Woodland or forest	1 (14%)	3 (16%)	4 (15%)
	National park	1	0	1
Taxon	Birds	3 (43%)	9 (47%)	12 (46%)
	Mammals	2 (29%)	9 (47%)	11 (42%)
	Vascular plants	2 (29%)	6 (32%)	8 (31%)
	Amphibians	1 (14%)	7 (37%)	8 (31%)
	Reptiles	1 (14%)	7 (37%)	8 (31%)
	Plants	0 (0%)	8 (42%)	8 (25%)
	General organisms	3 (43%)	2 (11%)	5 (19%)
	Moss	0 (0%)	2 (11%)	2 (8%)
	Invertebrates	1 (14%)	1 (5%)	2 (8%)
	the others	12	4	16

Note: [a] Frequency in number and percentage are shown in front of and in brackets, respectively. [b] Not applicable means the type of method doesn't apply the indicator. One method could cover one or more indicator values.

SPATIAL SCALE AND BIODIVERSITY REPRESENTATION

Biodiversity has high spatial heterogeneity. Spatial coverage is a crucial dimension related to a method's applicability, which decides where the method can be applied. Approximately, 46% of reviewed methods apply to a global scale (e.g., Mean Species Abundance (MSA)[43], the BioImpact Metric (BM)[44], Naturalness Degradation Indicator (NDI)[45], etc.) while the rest were developed at the continent, subcontinent, country (Swiss Agricultural LCA—Biodiversity (SALCA-BD)[27] for Switzerland, Expected Increase in the Number of Extinction Species-land use change (EINES-LUC)[46] for Japan, etc.), or a sub-country scale (e.g., Spectral Heterogeneity (SH))[41] (Fig.3). Methods with small spatial coverages are useless in a typical LCA study with global supply chains. Extending results from one region to others presents a challenge due to significant variations in biodiversity across regions[35]. Data availability is the main factor limiting the expansion of method coverage.

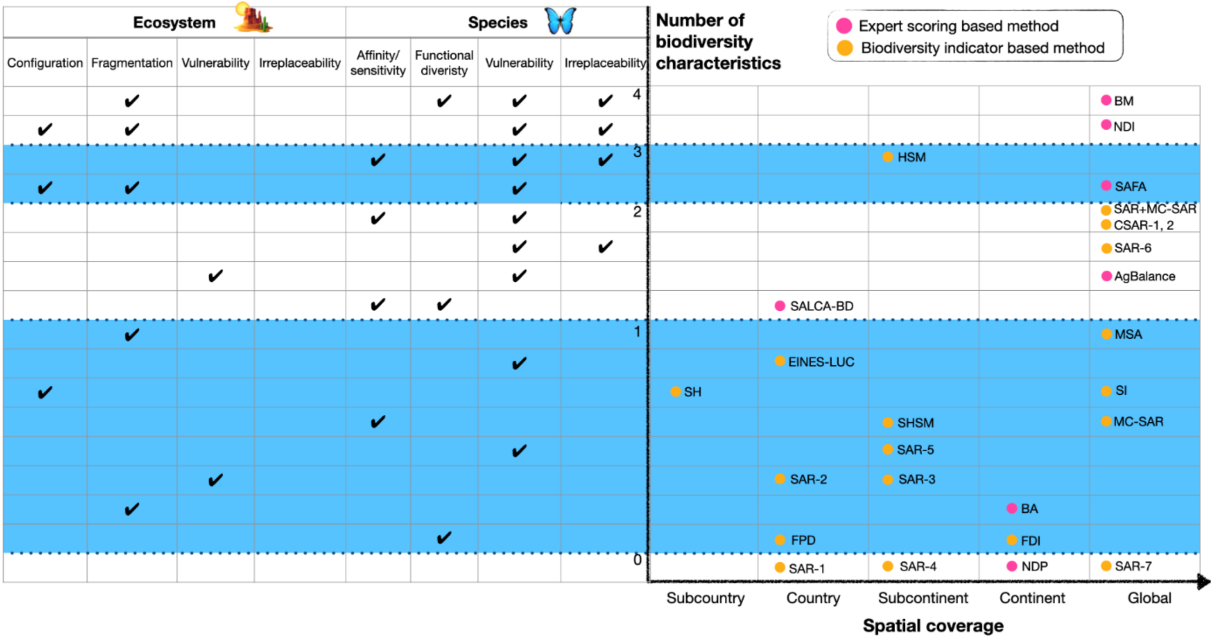


Fig.3 The number of biodiversity characteristics included (shown as ticks on the left) and spatial coverage of each method. Note: The Y-axis has 5 discrete grades; methods located within the same colour area have the same grade (the number of biodiversity characteristics included). Ticks in the left figure indicate the biodiversity characteristics used by the methods in the right figure within the same row. The more indicators ticked, the higher the biodiversity representation. Indicators of biodiversity characteristics were adapted from Curran, et al. [24]

The ecological values (or importance) vary among species depending on their characteristics. Thus, considering species vulnerability, habitat fragmentation, functional biodiversity, etc.[47, 48] in LCA methods would better represent biodiversity at the ecosystem and species levels. Initially, LCA methods for evaluating the impacts of biodiversity did not reflect the relative importance of different species. Recently, some studies captured several biodiversity characteristics at the species level. Species vulnerability is the most considered characteristic, followed by species irreplaceability and fragmentation (Fig.3). However, ecosystem irreplaceability was considered the least, followed by landscape configuration, ecosystem vulnerability, and functional diversity. Most of the methods involved 1-2 biodiversity characteristics. Two ESB methods considered the highest number of characteristics (BioImpact Metric (BM))[44] and Naturalness Degradation Indicator (NDI)[45]), but only four characteristics for each method. Therefore, it is hard to simultaneously measure biodiversity complexity in the Rio Conventions' broadest sense.

Some methods integrated species irreplaceability and vulnerability into the biodiversity impact assessment, such as SAR[15, 49], Naturalness Degradation Indicator (NDI)[45], Sustainability Assessment of Food and Agriculture Systems (SAFA)[50], etc. (Fig.3). Besides, some methods also considered species affinity (or suitability). One of the original SAR model's drawbacks is that it initially focused on natural habitats and assumed that no species persist in artificial habitats (including farmlands), which is not necessarily true[51]. The SAR-derived models, Countryside SAR (CSAR) [16, 52-54], or Matrix-calibrated SAR (MC-SAR)[35], can make up for the shortcomings of the SAR model. The Swiss Agricultural LCA—Biodiversity (SALCA-BD) method can achieve this goal with experts' judgment[27, 28]. As for functional biodiversity, it was considered in five of the reviewed methods, three ESB methods (Biodiversity Intactness Index (BII)[55], The BioImpact Metric (BM)[44], and Swiss Agricultural LCA—Biodiversity (SALCA-BD)[27, 28]) and two BIB methods (Functional Diversity Index (FDI)[47] and Functional Plant Diversity (FPD)[56]). From this perspective, ESB methods demonstrated a greater capacity to incorporate broader biodiversity characteristics through expert opinions. However, it should be noted that the species' vulnerability, irreplaceability, affinity, and functional data only covered limited species or areas[16, 47], which has limited the ability of biodiversity impact methods to include more species, landscape coverage, and biodiversity aspects.

A few methods have evaluated the biodiversity impacts at the ecosystem level, e.g., Satoyama Index (SI)[40] and Spectral heterogeneity (SH) methods[41]. Landscape fragmentation can affect biodiversity substantially. The metapopulation model has been developed in the biodiversity conservation area to estimate the extinction risk caused by land fragmentation[57]. However, the metapopulation model neglected the effects of matrix structure and quality on species movement through the landscape, which can significantly bias the results[58]. Besides, it relies on biologically detailed information seldom available for broad-scale assessments. Therefore, the network analysis[59] was applied to assess the landscape fragmentation, e.g., a recent method from Scherer, et al. [60] Landscape configuration is also recommended in LCA methods for biodiversity[61], which is taken into account by several methods (Habitat Suitability Models (HSM)[62], Satoyama Index (SI)[40], and Spectral heterogeneity (SH)[41]). However, the relationship between diversity and landscape configuration is still uncertain[63]. Landscapes with simpler configurations might support a higher diversity if the remaining habitats are in larger patches[64], while a more complex configuration might support higher biodiversity than simpler landscapes[65]. Similar to the biodiversity impact analysis at the species level, the ecological value of ecosystems could also be differentiated by their vulnerability and irreplaceability, as in AgBalance[26], and SAR[66, 67]. This paper selected eight biodiversity characteristics at the species and ecosystem levels (Fig.3). However, none of the methods covered the eight characteristics since LCA methods should strike a balance between applicability and complexity. Including more biodiversity characteristics, ideally, all of them would better reflect biodiversity's multidimensionality, but it is challenging.

BIODIVERSITY IMPACT ASSESSMENT OF LAND MANAGEMENT CHANGE

Current LCA methods for biodiversity impact analysis integrate land management change by applying land management intensity level (LMIL), land management practices (LMP), or a combination of them (Table 3). ESB methods mainly applied LMP, while BIB methods mainly applied the LMIL approach. LMIL is based on land classification systems and classifies a land use class into discrete intensity levels, e.g., organic versus conventional, extensive versus intensive, etc. [68]. In comparison, LMP accounts for various land management practices, e.g., pesticide application, fertilisation, ploughing, residue control, etc., by SALCA-BD[27, 28], cropping diversity, nitrogen surplus, etc., by AgBalance[26] (Table 3). However, for LMIL and LMP, no widespread consensus exists on how many and what land use intensity levels and land management practices should be considered for biodiversity impact analysis with LCA. Currently, diverse land use intensity levels and land management practices are used in different methods (Table 3).

Pros and cons exist in LMIL and LMP. LMIL can simplify the evaluation (e.g., inventory analysis phase), but it can only capture the differences in biodiversity impacts among those discrete land-use classes. The impact of different practices on biodiversity within the same land-use class will be diminished or negated (most BIB methods). LMP allows higher precision but increases complications. Applying LMP can also include some practices with positive impacts on biodiversity, e.g., AgBalance[26], Sustainability Assessment of Food and Agriculture Systems (SAFA)[50], etc. Land management practices and their impacts can vary widely by location and over time, making it challenging to evaluate their biodiversity impacts (most ESB methods). Therefore, several methods are considered to combine both approaches, e.g., NDP67 (Table 3), by considering several practices in classifying land-use classes, e.g., livestock intensity, fertilisation, etc. However, to make a compromise between LMIL and LMP, the land use intensity should be developed systematically.

Land management intensity includes several dimensions: 1) input intensity, 2) output intensity, and 3) changes in system properties, e.g., the complexity of ecosystems, carbon sequestration, etc.[69] The third dimension is related to many agroecological practices, e.g., ecological landscape design, hedgerows, flower strips, agroforestry, rice-fish coculture, etc., which affect the ecological patterns and processes. However, the multi-dimensional property is insufficiently reflected in biodiversity research[7], which mostly focuses on the first two dimensions. In the Land System (a land use classification system), land use intensity was represented by the efficiency of agricultural production (livestock density and yield gap)[20]. Chaudhary and Brooks [53] applied this method and developed land use intensity-specific characterization factors to evaluate biodiversity footprints at the global scale. Maskell, et al. [70] used the proportion of semi-natural and improved land to describe land management intensity. Tuck, et al. [71] evaluated land use intensity using arable field percentages. Maier, et al. [72] provided a methodological framework to evaluate land use intensity for different land use types by considering mostly agricultural inputs. A recent study developed characterization factors for the biodiversity impacts of land use intensities by considering specific land management practices[60], the land use intensity was defined by phosphorus and nitrogen inputs and the area equipped for irrigation[73]. Thus, a widespread method is yet to be established for evaluating land management intensity considering biodiversity impacts.

Table 3 The integration types of land management in different methods

Types	Method	LMIL and reference / LMP indicators	Land management-related ecosystem [c]
LMIL[a]	SAR-1[14]	Organic, less intensive, intensive. (Countryside Survey[74])	Agricultural areas, forests and semi-natural areas
	SAR-2[66]	Less intensive, intensive, etc. (CORINE land cover[75])	Agricultural areas, forests and semi-natural areas
	SAR-3[67]	Organic, less intensive, intensive, etc. (CORINE land cover[75])	Artificial surfaces, agricultural areas
	SAR-4[76]	Organic, conventional, etc. (CORINE land cover[75]; Countryside Survey[74])	Agricultural areas
	SAR-5[77]	Organic, less intensive, intensive, etc. (CORINE land cover[75])	Artificial surfaces, agricultural areas, forests and semi-natural areas
	SAR-6[78]	Organic, conventional.	Agricultural areas
	SAR-7[79]	Annual crops, permanent crops, agroforestry.	Agricultural areas, forests, artificial surfaces
	CSAR-1[53]	Minimal use, light use, intense use. (Global Land System[20])	Artificial surfaces, agricultural areas, forests and semi-natural areas
	CSAR-2[52]	Intensive, extensive. (Koellner et al. [68])	Forest and semi-natural areas
	MC-SAR[35]	Virgin and protected, management; low livestock density, high livestock density, etc. (LADA[80]; Anthromes[81])	Agricultural areas, forests and semi-natural areas, wetlands
	SAR+MC-SAR[82]	Annual crops, permanent crops.	Agricultural areas, forests and semi-natural areas, wetlands
	SHSM[83]	Deciduous orchard, evergreen orchard; irrigated grain crops, irrigated hayfield; rice, etc.[84]	Artificial surfaces, agricultural areas, forests and semi-natural areas, wetlands
	HSM[62]	Irrigated cropland, rainfed cropland, mosaic cropland, etc. (GlobCover v2.3[85])	Artificial surfaces, agricultural areas, forests and semi-natural areas, water bodies
	SI[40]	Cropland; rice paddies; other cropland.	Agricultural areas
	SH[41]	Vineyard, other crops.	Agricultural areas
	FDI[47]	Natural, used/artificial; extensive, intensive; etc.[68]	Agricultural areas, forests and semi-natural areas
	FPD[56]	Non-irrigated arable land; Pasture; Complex cultivation pattern. (CORINE land cover[75])	Agricultural areas, forests
	EINES-LUC[46]	Rice fields, grassland, other agricultural land[86].	Agricultural areas, forests, wetland
	MSA[43]	Low input, intensive, etc. (GLC 2000[87])	Agricultural areas, forests and semi-natural areas
LMP [b]	SALCA-BD[27]	Crop protection, fertilisation; crop rotation, soil cultivation, harvesting; cutting, grazing; conservation- crop margin, fallows (rotation), etc.	Agricultural areas
	AgBalance*[26]	Cropping diversity, nitrogen surplus, farming intensity, outcrossing potential, protected areas, ecotoxicity potential of pesticides.	Agricultural areas
	SAFA*[50]	Ecosystem enhancing practices; species conservation practices; wild genetic diversity enhancing practices	Agricultural areas

Types	Method	LMIL and reference / LMP indicators	Land management-related ecosystem [c]
	NDI[45]	Mechanical earth working, liming and fertilisation, pesticide deployment, the intensity of management interventions, etc.	Agricultural areas, forests and semi-natural areas
	BM*[44]	Not explicitly indicated	Agricultural areas, forests and semi-natural areas
	BA[88]	Nitrogen load	Artificial surfaces, agricultural areas, forest and semi-natural areas, wetlands, water bodies
LMIL+ LMP	NDP[89]	**LMIL:** Extensive, medium intensity, etc.[89]; **LMP:** Livestock intensity, ploughing, cutting, fertilisation, pesticides, rotation.	Artificial surfaces, agricultural areas, forest and semi-natural areas, wetlands

Note: [a] LMIL represents land management intensity level; [b] LMP means land management practices; [c] This classification is based on CORINE Land Cover[77]; * means some of the land management practices including positive biodiversity impacts; the methods in gray-shaded rows belong to Expert Scoring-Based method while the others belong to Biodiversity Indicator-Based method.

Biodiversity impact pathways of land management

Land management changes impact biodiversity directly (on-farm) and indirectly (off-farm) (Fig.4). Currently, operational LCA methods capture many indirect impacts through assessments of climate change, pollution, etc.[17], the direct impacts of on-farm biodiversity are often overlooked. Organic agriculture, for example, supports approximately 30% higher on-farm biodiversity compared to conventional agriculture[71], though it has approximately 25% lower yields globally with a variation among products and where they are produced[90]. The lower yields of organic agriculture may lead to agricultural land expansion[91] and higher biodiversity impacts per unit yield compared to conventional agriculture. However, organic agriculture can feed the world with lower N-surplus and pesticide, less land use than the reference scenario, if actions are taken from the food system, e.g., by reducing food waste, food-feed competition, reduced consumption and production of animal products[92]. Capturing the on-farm biodiversity impacts is crucial[93] as many agroecological practices can create synergies between yield and biodiversity[94]. These practices not only help maintain biodiversity but also mitigate long-term biodiversity loss outside natural protected areas[95], particularly on-farm[96]. Agroecological practices can reduce agricultural inputs, i.e., pollution, through biological nitrogen fixation, increasing the natural enemies, etc., and indirectly benefit biodiversity. They also directly enhance on-farm biodiversity; for example, perennial flower strips in apple orchards promote natural enemies[97], field margin floral enhancements (including hedgerows) increase pollinator abundance and richness[98], reduced tillage improves soil micro- and mesofauna densities compared to long-term intensive cultivation[99]. Therefore, assessing biodiversity impacts requires considering the positive effects of agroecological practices. The chosen reference situation substantially influences positive biodiversity impacts; a biodiversity impact value higher than the reference situation indicates a positive impact. However, under the same reference situation, positive biodiversity impacts from agroecological practices can be an added benefit to adjust the biodiversity impact value compared to the conventional systems that do not apply these practices. If the positive impacts of ecological practices on biodiversity are not considered, it will not be easy to incentivise responsible production and consumption. Thus, LCA studies of biodiversity impacts need to integrate positive biodiversity impacts of agroecological practices.

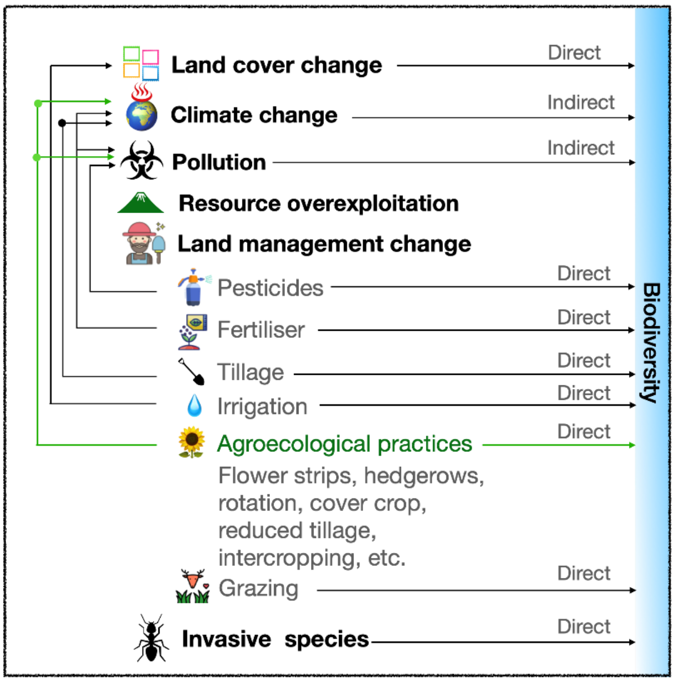


Fig. 4 Direct and indirect biodiversity impacts of biodiversity loss drivers, including land management change. Note: The green arrows represent the reduction of the environmental impacts and the positive impacts on biodiversity.

Biodiversity impacts from many land management practices, particularly agroecological practices, are rooted in ecological processes. To better capture these impacts, LCA methods need to incorporate cause-effect chains that account for these ecological processes. However, the current LCA methods focus on the cause-effect chains driven by extractions and emissions from material and energy flows[12]. For example, the impacts of pesticides are currently evaluated through environmental ecotoxicity, e.g., 1,4-DCB eq. (Recipe 2016 [100]) or potentially affected fraction in freshwater (USEtox[101]). However, selective pesticides can kill pollinators, birds, plants, microbes, or soil fauna. It will disrupt the on-farm insect trophic chain by killing a specific species and may result in species loss downstream of the trophic chain. Besides, the impacts of fertilisation on biodiversity have been captured through climate change and pollution (eutrophication, acidification, etc.). However, the direct impacts of fertilisation on biodiversity loss are also noteworthy. Fertilisation strongly reduced vascular plant species richness, shifted functional composition, and promoted nitrophilous species in mountain grasslands[102], due to competition for light because the sensitivity to the fertiliser varied among species[103]. Soil fauna diversity responded differently to fertilisation, depending on the application rate, soil and climatic conditions, and species in agroecosystems and temperate and boreal forest ecosystems[104]. Also, tillage[105], irrigation[106], and grazing[22] significantly affect biodiversity through ecological processes. Therefore, LCA methods should incorporate ecological process-based cause-effect chains to accurately reflect the biodiversity impacts of land management practices. This approach ensures no overlap with the extraction- and emission-driven cause-effect chains, thereby avoiding double counting.

RESEARCH NEEDS FOR THE FUTURE STUDIES

When evaluating the biodiversity impacts of agriproducts, it is crucial to consider land management change as a biodiversity loss driver. Potentially, it can be achieved by integrating the strengths of both ESB and BIB methods since they are complementary in many aspects. Most BIB methods can be applied globally, but they offer limited information on land management practices (Table 3, Table S3). In contrast, most ESB methods focus on land management change but are limited to the foreground systems [107] and cannot be applied to the supply chain and at a global scale (Table 3, Fig.3). Compared to BIB methods, ESB methods can include positive biodiversity impacts of agroecological practices and encompass a broader range of taxa and incorporate more biodiversity aspects at both the species and ecosystem levels (Table 2, Fig.3). ESB methods are often overly complex, whereas BIB methods tend to oversimplify in terms of inventory analysis. There are generally five inventory levels of the reviewed methods (Table S3): Specific crop type (Level 1), Crop category (Level 2), Organic and conventional (Level 3), Land use intensity (Level 4), and Land management practices (Level 5). As the complexity of the analysis increases from Level 1 to Level 5, focusing on land-use intensity based on management practices offers a middle ground between simplicity and complexity. This approach strikes a balance over-simplification and over-complication in the inventory analysis phase. To effectively integrate ESB and BIB methods, it is recommended to characterise land-use intensity based on specific land management practices and include the positive impact of agroecological practices.

Future research needs to establish a land use intensity evaluation method based on land management practices by considering the three land management intensity dimensions, i.e., input intensity, output intensity, and especially the changes in system properties that is related to the agroecological practices [69]. Data availability is one of the main challenges in integrating biodiversity impacts into LCA globally. Statistical or expert estimates can be used for inputs and output intensity dimensions, e.g., fertiliser types, nutrient inputs, pesticide inputs, etc. As for the dimension of changes in system properties, leveraging remote sensing makes land management practice data more available[108], e.g., agroforestry[109], rotation[110], intercropping[111], etc. Many of the agroecological practices can have positive impacts on biodiversity. Positive biodiversity impacts are context-specific, making them difficult to standardise across different regions, ecosystems, and agroecological practices. Recently, Bonfanti, et al. [112] established a global database of the impacts of agricultural management practices on terrestrial biodiversity, encompassing 8 primary individual field practices, 3 agricultural systems, and 2 landscape-level interventions. Building on this global database, it is promising to create characterisation factors that can precisely capture positive biodiversity impacts of land management practices. The other challenge is to rigorously establish quantitative relationships between land use intensity and biodiversity impacts. Lindner, et al. [113] applied fuzzy thinking to assess the biodiversity impacts of forest land management practices based on expert opinion, which could also be applied to the agroecosystems.

Author's statement

Huayang Zhen: Conceptualization, Methodology, Writing-original draft, Writing-review and editing; **Pietro Goglio**: Conceptualization, Writing-review and editing, project administration, Funding Acquisition, Supervision; **Fatemeh Hashemi, Maxime Fossey, and Christel Cederberg**: Writing-review and editing; **Marie Trydeman Knudsen**: Conceptualization, Writing-review and editing, project administration, Funding Acquisition, Supervision.

Declaration of competing interest

None.

Acknowledgments

This research was funded by the PATHWAYS project (Grant No. 101000395) and partly funded by the MIXED project (Grant No. 862357) under the HORIZON 2020 program, and the Ministry of Food, Agriculture & Fisheries of Denmark (Grant No. 41932 LBST NIFA MATK). The authors wish to acknowledge Dr. Hayo van der Werf for his precious comments and insights.

References

(1) Brauman, K. A.; Garibaldi, L. A.; Polasky, S.; Aumeeruddy-Thomas, Y.; Brancalion, P. H.; DeClerck, F.; Jacob, U.; Mastrangelo, M. E.; Nkongolo, N. V.; Palang, H. Global trends in nature's contributions to people. *Proceedings of the National Academy of Sciences* **2020**, *117* (51), 32799-32805.

(2) Pimm, S. L.; Jenkins, C. N.; Abell, R.; Brooks, T. M.; Gittleman, J. L.; Joppa, L. N.; Raven, P. H.; Roberts, C. M.; Sexton, J. O. The biodiversity of species and their rates of extinction, distribution, and protection. *science* **2014**, *344* (6187), 1246752.

(3) Díaz, S.; Settele, J.; Brondízio, E. S.; Ngo, H. T.; Agard, J.; Arneth, A.; Balvanera, P.; Brauman, K. A.; Butchart, S. H.; Chan, K. M. Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science* **2019**, *366* (6471), eaax3100.

(4) FAO. *FAOSTAT*. 2022. (accessed 2022 31st December).

(5) Maxwell, S. L.; Fuller, R. A.; Brooks, T. M.; Watson, J. E. M. Biodiversity: The ravages of guns, nets and bulldozers. *Nature* **2016**, *536* (7615), 143-145.

(6) Karp, D. S.; Rominger, A. J.; Zook, J.; Ranganathan, J.; Ehrlich, P. R.; Daily, G. C. Intensive agriculture erodes beta-diversity at large scales. *Ecology Letters* **2012**, *15* (9), 963-970.

(7) Dullinger, I.; Essl, F.; Moser, D.; Erb, K.; Haberl, H.; Dullinger, S. Biodiversity models need to represent land-use intensity more comprehensively. *Global Ecology and Biogeography* **2021**, *30* (5), 924-932.

(8) Shukla, P. R.; Skeg, J.; Buendia, E. C.; Masson-Delmotte, V.; Pörtner, H.-O.; Roberts, D.; Zhai, P.; Slade, R.; Connors, S.; Van Diemen, S. Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems. **2019**.

(9) Harrison, P. A.; Berry, P. M.; Simpson, G.; Haslett, J. R.; Blicharska, M.; Bucur, M.; Dunford, R.; Egoh, B.; Garcia-Llorente, M.; Geamănă, N.; et al. Linkages between biodiversity attributes and ecosystem services: A systematic review. *Ecosystem Services* **2014**, *9*, 191-203.

(10) van der Werf, H. M. G.; Knudsen, M. T.; Cederberg, C. Towards better representation of organic agriculture in life cycle assessment. *Nature Sustainability* **2020**, *3* (6), 419-425.

(11) Wilting, H. C.; Schipper, A. M.; Bakkenes, M.; Meijer, J. R.; Huijbregts, M. A. J. Quantifying Biodiversity Losses Due to Human Consumption: A Global-Scale Footprint Analysis. *Environmental Science & Technology* **2017**, *51* (6), 3298-3306.

(12) ISO. *ISO 14044-Environmental Managementelife Cycle Assessment -Requirements and Guidelines*; 2006. ISO. *ISO 14040-Environmental Managemente Life Cycle Assessment -Principles and Framework*; 2006.

(13) Koellner, T.; De Baan, L.; Beck, T.; Brandão, M.; Civit, B.; Margni, M.; Saad, R.; de Souza, D. M.; Müller-Wenk, R. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *The International Journal of Life Cycle Assessment* **2013**, *18* (6), 1188-1202. Milà i Canals, L.; Bauer, C.; Depestele, J.; Dubreuil, A.; Freiermuth Knuchel, R.; Gaillard, G.; Michelsen, O.; Müller-Wenk, R.; Rydgren, B. Key elements in a framework for land use impact assessment within LCA. *The International Journal of Life Cycle Assessment* **2007**, *12* (1), 5-15.

(14) De Schryver, A. M.; Goedkoop, M. J.; Leuven, R. S.; Huijbregts, M. A. Uncertainties in the application of the species area relationship for characterisation factors of land occupation in life cycle assessment. *The International Journal of Life Cycle Assessment* **2010**, *15* (7), 682-691.

(15) Pezzati, L.; Verones, F.; Curran, M.; Baustert, P.; Hellweg, S. Biodiversity recovery and transformation impacts for wetland biodiversity. *Environmental Science & Technology* **2018**, *52* (15), 8479-8487.

(16) Chaudhary, A.; Verones, F.; De Baan, L.; Hellweg, S. Quantifying land use impacts on biodiversity: combining species–area models and vulnerability indicators. *Environmental Science & Technology* **2015**, *49* (16), 9987-9995.

(17) Crenna, E.; Marques, A.; La Notte, A.; Sala, S. Biodiversity assessment of value chains: state of the art and emerging challenges. *Environmental Science & Technology* **2020**, *54* (16), 9715-9728.

(18) Gabel, V. M.; Meier, M. S.; Köpke, U.; Stolze, M. The challenges of including impacts on biodiversity in agricultural life cycle assessments. *Journal of Environmental Management* **2016**, *181*, 249-260.

(19) Turner, B. L.; Lambin, E. F.; Reenberg, A. The emergence of land change science for global environmental change and sustainability. *Proceedings of the National Academy of Sciences* **2007**, *104* (52), 20666-20671.

(20) van Asselen, S.; Verburg, P. H. A land system representation for global assessments and land-use modeling. *Global Change Biology* **2012**, *18* (10), 3125-3148.

(21) Pretty, J. Intensification for redesigned and sustainable agricultural systems. *Science* **2018**, *362* (6417), eaav0294.

(22) Wang, C.; Tang, Y. A global meta-analyses of the response of multi-taxa diversity to grazing intensity in grasslands. *Environmental Research Letters* **2019**, *14* (11), 114003.

(23) Marques, A.; Robuchon, M.; Hellweg, S.; Newbold, T.; Beher, J.; Bekker, S.; Essl, F.; Ehrlich, D.; Hill, S.; Jung, M. A research perspective towards a more complete biodiversity footprint: a report from the World Biodiversity Forum. *The International Journal of Life Cycle Assessment* **2021**, *26* (2), 238-243.

(24) Curran, M.; Maia de Souza, D.; Antón, A.; Teixeira, R. F.; Michelsen, O.; Vidal-Legaz, B.; Sala, S.; Mila i Canals, L. How Well Does LCA Model Land Use Impacts on Biodiversity? A Comparison with Approaches from Ecology and Conservation. *Environmental Science & Technology* **2016**, *50* (6), 2782-2795.

(25) Damiani, M.; Sinkko, T.; Caldeira, C.; Tosches, D.; Robuchon, M.; Sala, S. Critical review of methods and models for biodiversity impact assessment and their applicability in the LCA context. *Environmental Impact Assessment Review* **2023**, *101*, 107134.

(26) Saling, P.; Schöneboom, J.; Künast, C.; Ufer, A.; Gipmans, M.; Frank, M. Assessment of Biodiversity within the Holistic Sustainability Evaluation Method of AgBalance. In *9th International Conference LCA of Food*, 2014.

(27) Jeanneret, P.; Baumgartner, D. U.; Knuchel, R. F.; Koch, B.; Gaillard, G. An expert system for integrating biodiversity into agricultural life-cycle assessment. *Ecological Indicators* **2014**, *46*, 224-231.

(28) Lüscher, G.; Nemecek, T.; Arndorfer, M.; Balázs, K.; Dennis, P.; Fjellstad, W.; Friedel, J. K.; Gaillard, G.; Herzog, F.; Sarthou, J.-P. Biodiversity assessment in LCA: a validation at field and farm scale in eight European regions. *The International Journal of Life Cycle Assessment* **2017**, *22* (10), 1483-1492.

(29) Penman, T. D.; Law, B. S.; Ximenes, F. A proposal for accounting for biodiversity in life cycle assessment. *Biodiversity and Conservation* **2010**, *19* (11), 3245-3254.

(30) Moher, D.; Liberati, A.; Tetzlaff, J.; Altman, D. G.; PRISMA Group*, t. Preferred reporting items for systematic reviews and meta-analyses: the PRISMA statement. *Annals of internal medicine* **2009**, *151* (4), 264-269.

(31) Pongratz, J.; Dolman, H.; Don, A.; Erb, K. H.; Fuchs, R.; Herold, M.; Jones, C.; Kuemmerle, T.; Luyssaert, S.; Meyfroidt, P. Models meet data: Challenges and opportunities in implementing land management in Earth system models. *Global change biology* **2018**, *24* (4), 1470-1487.

(32) IPCC. *IPCC Guidelines for National Greenhouse Gas Inventories*; Japan, 2006.

(33) Goglio, P.; Knudsen, M. T.; Van Mierlo, K.; Röhrig, N.; Fossey, M.; Maresca, A.; Hashemi, F.; Waqas, M. A.; Yngvesson, J.; Nassy, G.; et al. Defining common criteria for harmonizing life cycle assessments of livestock systems. *Cleaner Production Letters* **2023**, *4*, 100035.

(34) FAO. *Biodiversity and the livestock sector-Guidelines for quantitative assessment-Version 1*; FAO LEAP, ROME, 2020. DOI: http://www.fao.org/partnerships/leap/en/.

(35) De Baan, L.; Mutel, C. L.; Curran, M.; Hellweg, S.; Koellner, T. Land use in life cycle assessment: global characterization factors based on regional and global potential species extinction. *Environmental Science & Technology* **2013**, *47* (16), 9281-9290.

(36) Curran, M.; de Baan, L.; De Schryver, A. M.; Van Zelm, R.; Hellweg, S.; Koellner, T.; Sonnemann, G.; Huijbregts, M. A. Toward meaningful end points of biodiversity in life cycle assessment. *Environmental Science & Technology* **2011**, *45* (1), 70-79.

(37) Hunt, M. L.; Blackburn, G. A.; Rowland, C. S. Monitoring the Sustainable Intensification of Arable Agriculture: the Potential Role of Earth Observation. *International Journal of Applied Earth Observation and Geoinformation* **2019**, *81*, 125-136.

(38) Garrigues, E.; Corson, M. S.; Angers, D. A.; van der Werf, H. M. G.; Walter, C. Soil quality in Life Cycle Assessment: Towards development of an indicator. *Ecological Indicators* **2012**, *18*, 434-442.

(39) Vrasdonk, E.; Palme, U.; Lennartsson, T. Reference situations for biodiversity in life cycle assessments: conceptual bridging between LCA and conservation biology. *International Journal of Life Cycle Assessment* **2019**, *24* (9), 1631-1642.

(40) Kadoya, T.; Washitani, I. The Satoyama Index: A biodiversity indicator for agricultural landscapes. *Agriculture, Ecosystems & Environment* **2011**, *140* (1-2), 20-26.

(41) Rugani, B.; Rocchini, D. Boosting the use of spectral heterogeneity in the impact assessment of agricultural land use on biodiversity. *Journal of Cleaner Production* **2017**, *140*, 516-524.

(42) Purvis, A.; Hector, A. Getting the measure of biodiversity. *Nature* **2000**, *405* (6783), 212-219.

(43) Alkemade, R.; Van Oorschot, M.; Miles, L.; Nellemann, C.; Bakkenes, M.; Ten Brink, B. GLOBIO3: a framework to investigate options for reducing global terrestrial biodiversity loss. *Ecosystems* **2009**, *12* (3), 374-390.

(44) Turner, P. A.; Ximenes, F. A.; Penman, T. D.; Law, B. S.; Waters, C. M.; Grant, T.; Mo, M.; Brock, P. M. Accounting for biodiversity in life cycle impact assessments of forestry and agricultural systems—the BioImpact metric. *The International Journal of Life Cycle Assessment* **2019**, *24* (11), 1985-2007.

(45) Fehrenbach, H.; Grahl, B.; Giegrich, J.; Busch, M. Hemeroby as an impact category indicator for the integration of land use into life cycle (impact) assessment. *The International Journal of Life Cycle Assessment* **2015**, *20* (11), 1511-1527.

(46) Yamaguchi, K.; Ii, R.; Itsubo, N. Ecosystem damage assessment of land transformation using species loss. *The International Journal of Life Cycle Assessment* **2018**, *23* (12), 2327-2338.

(47) de Souza, D. M.; Flynn, D. F.; DeClerck, F.; Rosenbaum, R. K.; de Melo Lisboa, H.; Koellner, T. Land use impacts on biodiversity in LCA: proposal of characterization factors based on functional diversity. *The International Journal of Life Cycle Assessment* **2013**, *18* (6), 1231-1242.

(48) Souza, D. M.; Teixeira, R. F.; Ostermann, O. P. Assessing biodiversity loss due to land use with Life Cycle Assessment: are we there yet? *Global change biology* **2015**, *21* (1), 32-47.

(49) Verones, F.; Pfister, S.; Hellweg, S. Quantifying area changes of internationally important wetlands due to water consumption in LCA. *Environmental Science & Technology* **2013**, *47* (17), 9799-9807.

(50) Hanisch, A. L.; Negrelle, R. R.; Bonatto, R. A.; Nimmo, E. R.; Lacerda, A. E. B. Evaluating sustainability in traditional silvopastoral systems (caívas): looking beyond the impact of animals on biodiversity. *Sustainability* **2019**, *11* (11), 3098.

(51) Pereira, H. M.; Daily, G. C. Modeling biodiversity dynamics in countryside landscapes. *Ecology* **2006**, *87* (8), 1877-1885.

(52) Chaudhary, A.; Verones, F.; De Baan, L.; Hellweg, S. Quantifying land use impacts on biodiversity: combining species–area models and vulnerability indicators. *Environmental Science & Technologyy* **2015**, *49* (16), 9987-9995.

(53) Chaudhary, A.; Brooks, T. M. Land use intensity-specific global characterization factors to assess product biodiversity footprints. *Environmental Science & Technology* **2018**, *52* (9), 5094-5104.

(54) Dorber, M.; Kuipers, K.; Verones, F. Global characterization factors for terrestrial biodiversity impacts of future land inundation in Life Cycle Assessment. *Science of The Total Environment* **2020**, *712*, 134582. Chaudhary, A.; Carrasco, L. R.; Kastner, T. Linking national wood consumption with global biodiversity and ecosystem service losses. *Science of The Total Environment* **2017**, *586*, 985-994.

(55) Scholes, R. J.; Biggs, R. A biodiversity intactness index. *Nature* **2005**, *434* (7029), 45-49.

(56) Scherer, L.; van Baren, S. A.; van Bodegom, P. M. Characterizing Land Use Impacts on Functional Plant Diversity for Life Cycle Assessments. *Environmental Science & Technology* **2020**, *54* (11), 6486-6495.

(57) Schnell, J. K.; Harris, G. M.; Pimm, S. L.; Russell, G. J. Estimating extinction risk with metapopulation models of large-scale fragmentation. *Conservation Biology* **2013**, *27* (3), 520-530. Hanski, I.; Zurita, G. A.; Bellocq, M. I.; Rybicki, J. Species–fragmented area relationship. *Proceedings of the National Academy of Sciences* **2013**, *110* (31), 12715-12720.

(58) Larrey-Lassalle, P.; Esnouf, A.; Roux, P.; Lopez-Ferber, M.; Rosenbaum, R. K.; Loiseau, E. A methodology to assess habitat fragmentation effects through regional indexes: illustration with forest biodiversity hotspots. *Ecological Indicators* **2018**, *89*, 543-551.

(59) Saura, S.; Estreguil, C.; Mouton, C.; Rodríguez-Freire, M. Network analysis to assess landscape connectivity trends: Application to European forests (1990–2000). *Ecological Indicators* **2011**, *11* (2), 407-416.

(60) Scherer, L.; Rosa, F.; Sun, Z.; Michelsen, O.; De Laurentiis, V.; Marques, A.; Pfister, S.; Verones, F.; Kuipers, K. J. J. Biodiversity Impact Assessment Considering Land Use Intensities and Fragmentation. *Environmental Science & Technology* **2023**, *57* (48), 19612-19623.

(61) Teixeira, R. F.; de Souza, D. M.; Curran, M. P.; Antón, A.; Michelsen, O.; i Canals, L. M. Towards consensus on land use impacts on biodiversity in LCA: UNEP/SETAC Life Cycle Initiative preliminary recommendations based on expert contributions. *Journal of Cleaner Production* **2016**, *112*, 4283-4287.

(62) de Baan, L.; Curran, M.; Rondinini, C.; Visconti, P.; Hellweg, S.; Koellner, T. High-resolution assessment of land use impacts on biodiversity in life cycle assessment using species habitat suitability models. *Environmental Science & Technology* **2015**, *49* (4), 2237-2244.

(63) Seppelt, R.; Beckmann, M.; Ceauşu, S.; Cord, A. F.; Gerstner, K.; Gurevitch, J.; Kambach, S.; Klotz, S.; Mendenhall, C.; Phillips, H. R. Harmonizing biodiversity conservation and productivity in the context of increasing demands on landscapes. *BioScience* **2016**, *66* (10), 890-896.

(64) Gerstner, K.; Dormann, C. F.; Václavík, T.; Kreft, H.; Seppelt, R. Accounting for geographical variation in species–area relationships improves the prediction of plant species richness at the global scale. *Journal of Biogeography* **2014**, *41* (2), 261-273.

(65) Stein, A.; Gerstner, K.; Kreft, H. Environmental heterogeneity as a universal driver of species richness across taxa, biomes and spatial scales. *Ecology Letters* **2014**, *17* (7), 866-880.

(66) Schmidt, J. H. Development of LCIA characterisation factors for land use impacts on biodiversity. *Journal of Cleaner Production* **2008**, *16* (18), 1929-1942.

(67) Köllner, T. Species-pool effect potentials (SPEP) as a yardstick to evaluate land-use impacts on biodiversity. *Journal of cleaner production* **2000**, *8* (4), 293-311.

(68) Koellner, T.; De Baan, L.; Beck, T.; Brandão, M.; Civit, B.; Goedkoop, M.; Margni, M.; Müller-Wenk, R.; Weidema, B.; Wittstock, B. Principles for life cycle inventories of land use on a global scale. *The International Journal of Life Cycle Assessment* **2013**, *18* (6), 1203-1215.

(69) Erb, K.-H.; Haberl, H.; Jepsen, M. R.; Kuemmerle, T.; Lindner, M.; Müller, D.; Verburg, P. H.; Reenberg, A. A conceptual framework for analysing and measuring land-use intensity. *Current opinion in environmental sustainability* **2013**, *5* (5), 464-470.

(70) Maskell, L. C.; Botham, M.; Henrys, P.; Jarvis, S.; Maxwell, D.; Robinson, D. A.; Rowland, C. S.; Siriwardena, G.; Smart, S.; Skates, J.; et al. Exploring relationships between land use intensity, habitat heterogeneity and biodiversity to identify and monitor areas of High Nature Value farming. *Biological Conservation* **2019**, *231*, 30-38.

(71) Tuck, S. L.; Winqvist, C.; Mota, F.; Ahnström, J.; Turnbull, L. A.; Bengtsson, J. Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology* **2014**, *51* (3), 746-755.

(72) Maier, S. D.; Lindner, J. P.; Francisco, J. Conceptual framework for biodiversity assessments in Global value chains. *Sustainability* **2019**, *11* (7), 1841.

(73) Newbold, T.; Hudson, L. N.; Hill, S. L. L.; Contu, S.; Lysenko, I.; Senior, R. A.; Börger, L.; Bennett, D. J.; Choimes, A.; Collen, B.; et al. Global effects of land use on local terrestrial biodiversity. *Nature* **2015**, *520* (7545), 45-50.

(74) Sheail, J.; Bunce, R. The development and scientific principles of an environmental classification for strategic ecological survey in the United Kingdom. *Environmental Conservation* **2003**, *30* (2), 147-159.

(75) CLMS; EEA. *CORINE Land Cover Product User Manual (Version 1.0)*; 2021. DOI: https://land.copernicus.eu/user-corner/technical-library/clc-product-user-manual.

(76) Knudsen, M. T.; Hermansen, J. E.; Cederberg, C.; Herzog, F.; Vale, J.; Jeanneret, P.; Sarthou, J.-P.; Friedel, J. K.; Balázs, K.; Fjellstad, W. Characterization factors for land use impacts on biodiversity in life cycle assessment based on direct measures of plant species richness in European farmland in the 'Temperate Broadleaf and Mixed Forest'biome. *Science of the Total Environment* **2017**, *580*, 358-366.

(77) Koellner, T.; Scholz, R. W. Assessment of land use impacts on the natural environment. *The International Journal of Life Cycle Assessment* **2008**, *13* (1), 32-48.

(78) Mueller, C.; de Baan, L.; Koellner, T. Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study. *The International Journal of Life Cycle Assessment* **2014**, *19* (1), 52-68.

(79) de Baan, L.; Alkemade, R.; Koellner, T. Land use impacts on biodiversity in LCA: a global approach. *The International Journal of Life Cycle Assessment* **2013**, *18* (6), 1216-1230.

(80) Nachtergaele, F.; Petri, M. *Mapping land use systems at global and regional scales for land degradation assessment analysis (Version 1.1)*; GEF, UNEP, and FAO, Rome, 2013. DOI: https://www.fao.org/3/i3242e/i3242e.pdf.

(81) Ellis, E. C.; Ramankutty, N. Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment* **2008**, *6* (8), 439-447.

(82) Verones, F.; Huijbregts, M. A.; Chaudhary, A.; de Baan, L.; Koellner, T.; Hellweg, S. Harmonizing the assessment of biodiversity effects from land and water use within LCA. *Environmental Science & Technology* **2015**, *49* (6), 3584-3592.

(83) Geyer, R.; Lindner, J. P.; Stoms, D. M.; Davis, F. W.; Wittstock, B. Coupling GIS and LCA for biodiversity assessments of land use: Part 2: Impact assessment. *The International Journal of Life Cycle Assessment* **2010**, *15* (7).

(84) KE, M.; WF, L. *A guide to wildlife habitats of California*; State of California Resources Agency, Sacramento, 1988.

(85) ESA; UCL. *European Space Agency GlobCover land cover map, v2.3*. 2010. https://due.esrin.esa.int/page%5Fglobcover.php (accessed 2024 27 August).

(86) Japan, M. o. t. E. o.; (GIO), G. G. I. O. o. J.; CGER; NIES. *National Greenhouse Gas Inventory Report of Japan*; National Institute for Environmental Studies, Ibaraki, 2012.

(87) Bartholome, E.; Belward, A.; Achard, F.; Bartalev, S.; Carmona-Moreno, C.; Eva, H.; Fritz, S.; Gregoire, J.; Mayaux, P.; Stibig, H. GLC 2000: Global Land Cover mapping for the year 2000. *Project status, November* **2002**, *20*.

(88) Gardi, C.; Jeffery, S.; Saltelli, A. An estimate of potential threats levels to soil biodiversity in EU. *Global change biology* **2013**, *19* (5), 1538-1548.

(89) Brentrup, F.; Küsters, J.; Lammel, J.; Kuhlmann, H. Life cycle impact assessment of land use based on the hemeroby concept. *The International Journal of Life Cycle Assessment* **2002**, *7* (6), 339-348.

(90) Seufert, V.; Ramankutty, N.; Foley, J. A. Comparing the yields of organic and conventional agriculture. *Nature* **2012**, *485* (7397), 229-232.

(91) Gong, S.; Hodgson, J. A.; Tscharntke, T.; Liu, Y.; van der Werf, W.; Batáry, P.; Knops, J. M.; Zou, Y. Biodiversity and yield trade-offs for organic farming. *Ecology letters* **2022**, *25* (7), 1699-1710.

(92) Muller, A.; Schader, C.; El-Hage Scialabba, N.; Bruggemann, J.; Isensee, A.; Erb, K. H.; Smith, P.; Klocke, P.; Leiber, F.; Stolze, M.; et al. Strategies for feeding the world more sustainably with organic agriculture. *Nat Commun* **2017**, *8* (1), 1290.

(93) van der Werf, H. M. G.; Knudsen, M. T.; Cederberg, C. Towards better representation of organic agriculture in life cycle assessment. *Nature Sustainability* **2020**.

(94) Tamburini, G.; Bommarco, R.; Wanger, T. C.; Kremen, C.; Van Der Heijden, M. G.; Liebman, M.; Hallin, S. Agricultural diversification promotes multiple ecosystem services without compromising yield. *Science Advances* **2020**, *6* (45), eaba1715.

(95) Hendershot, J. N.; Smith, J. R.; Anderson, C. B.; Letten, A. D.; Frishkoff, L. O.; Zook, J. R.; Fukami, T.; Daily, G. C. Intensive farming drives long-term shifts in avian community composition. *Nature* **2020**, *579* (7799), 393-396. Kremen, C.; Merenlender, A. M. Landscapes that work for biodiversity and people. *Science* **2018**, *362* (6412), eaau6020.

(96) Aguilera, E.; Diaz-Gaona, C.; Garcia-Laureano, R.; Reyes-Palomo, C.; Guzmán, G. I.; Ortolani, L.; Sanchez-Rodriguez, M.; Rodriguez-Estevez, V. Agroecology for adaptation to climate change and resource depletion in the Mediterranean region. A review. *Agricultural Systems* **2020**, *181*, 102809.

(97) Jacobsen, S. K.; Sørensen, H.; Sigsgaard, L. Perennial flower strips in apple orchards promote natural enemies in their proximity. *Crop Protection* **2022**, *156*, 105962.

(98) Zamorano, J.; Bartomeus, I.; Grez, A. A.; Garibaldi, L. A. Field margin floral enhancements increase pollinator diversity at the field edge but show no consistent spillover into the crop field: a meta-analysis. *Insect Conservation and Diversity* **2020**, *13* (6), 519-531.

(99) Betancur-Corredor, B.; Lang, B.; Russell, D. J. Reducing tillage intensity benefits the soil micro-and mesofauna in a global meta-analysis. *AgriRxiv* **2022**, (2022), 20220266588.

(100) 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. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment* **2016**, *22* (2), 138-147.

(101) Rosenbaum, R. K.; Bachmann, T. M.; Gold, L. S.; Huijbregts, M. A.; Jolliet, O.; Juraske, R.; Koehler, A.; Larsen, H. F.; MacLeod, M.; Margni, M. USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. *The International Journal of Life Cycle Assessment* **2008**, *13* (7), 532-546.

(102) Boch, S.; Kurtogullari, Y.; Allan, E.; Lessard-Therrien, M.; Rieder, N. S.; Fischer, M.; De León, G. M.; Arlettaz, R.; Humbert, J.-Y. Effects of fertilization and irrigation on vascular plant species richness, functional composition and yield in mountain grasslands. *Journal of Environmental Management* **2021**, *279*, 111629.

(103) Hautier, Y.; Niklaus, P. A.; Hector, A. Competition for light causes plant biodiversity loss after eutrophication. *Science* **2009**, *324* (5927), 636-638.

(104) Sullivan, T. P.; Sullivan, D. S. Influence of nitrogen fertilization on abundance and diversity of plants and animals in temperate and boreal forests. *Environmental Reviews* **2018**, *26* (1), 26-42. Betancur-Corredor, B.; Lang, B.; Russell, D. J. Organic nitrogen fertilization benefits selected soil fauna in global agroecosystems. *Biology and Fertility of Soils* **2022**, 1-16.

(105) Rieff, G. G.; Natal-da-Luz, T.; Renaud, M.; Azevedo-Pereira, H. M.; Chichorro, F.; Schmelz, R. M.; de Sá, E. L. S.; Sousa, J. P. Impact of no-tillage versus conventional maize plantation on soil mesofauna with and without the use of a lambda-cyhalothrin based insecticide: A terrestrial model ecosystem experiment. *Applied Soil Ecology* **2020**, *147*, 103381.

(106) Cabodevilla, X.; Wright, A. D.; Villanua, D.; Arroyo, B.; Zipkin, E. F. The implementation of irrigation leads to declines in farmland birds. *Agriculture, Ecosystems & Environment* **2022**, *323*, 107701.

(107) Pépin, A.; Guidoboni, M. V.; Jeanneret, P.; van der Werf, H. M. Using an expert system to assess biodiversity in life cycle assessment of vegetable crops. *Ecological Indicators* **2023**, *148*, 110098.

(108) Bégué, A.; Arvor, D.; Bellon, B.; Betbeder, J.; de Abelleyra, D.; P. D. Ferraz, R.; Lebourgeois, V.; Lelong, C.; Simões, M.; R. Verón, S. Remote Sensing and Cropping Practices: A Review. *Remote Sensing* **2018**, *10* (2).

(109) Bolívar-Santamaría, S.; Reu, B. Detection and characterization of agroforestry systems in the Colombian Andes using sentinel-2 imagery. *Agrofor. Syst.* **2021**, *95* (3), 499-514.

(110) Peguero, G.; Burkart, A.; Íñiguez, E.; Rodríguez, A.; Llurba, R.; Sebastià, M. T. Remote sensing of legacy effects of biodiversity on crop performance. *Agriculture, Ecosystems & Environment* **2023**, *345*.

(111) Parra, L.; Mostaza-Colado, D.; Marin, J. F.; Mauri, P. V.; Lloret, J. Methodology to Differentiate Legume Species in Intercropping Agroecosystems Based on UAV with RGB Camera. *Electronics* **2022**, *11* (4).

(112) Bonfanti, J.; Langridge, J.; Beillouin, D. A global database to catalogue the impacts of agricultural management practices on terrestrial biodiversity. *Data in Brief* **2023**, *50*, 109555.

(113) Lindner, J. P.; Eberle, U.; Knuepffer, E.; Coelho, C. R. Moving beyond land use intensity types: assessing biodiversity impacts using fuzzy thinking. *The International Journal of Life Cycle Assessment* **2021**, *26* (7), 1338-1356.

SG3: Animal welfare

Review of recent approaches for the inclusion of animal welfare in Life Cycle Assessment (LCA)

Nina Adams[a], Jenny Yngvesson[b], Maxime Fossey[c], Manuel Romero Huelva[d], Harry Blokhuis[e], Daiana de Oliveira[e,f], Alberto Maresca[g], Greg Thoma[h], Andrea Vitali[i], Nicholas Davison[a], Laurence Smith[a,j] (*corresponding author*), Pietro Goglio[k]

[a] School of Agriculture, Policy and Development, University of Reading, Whiteknights, RG6 6EU, Reading, UK; ninaroehrig@live.de
[b] Department of Applied Animal Science and Welfare, Swedish University of Agricultural Sciences, Box 234, SE-532 23 Skara, Sweden
[c] Institut de l'élevage (IDELE), 149 rue de Bercy, 75012 Paris, France
[d] Estación Experimental del Zaidín (CSIC), Profesor Albareda 1, 18008 Granada, Spain
[e] Department of Applied Animal Science and Welfare, Swedish University of Agricultural Sciences, Box 7024, SE-750 07 Uppsala, Sweden
[f] Department of Biology and Environmental Science, Linnaeus University, SE-391 82 Kalmar, Sweden
[g] SEGES Innovation P/S, Agro Food Park 15, 8200 Aarhus, Denmark
[h] AgNext, Colorado State University, 350 West Pitkin Street, CO 80523 Fort Collins, USA
[i] Department of Agriculture and Forestry Sciences, University of Tuscia, Via San Camillo De Lellis, 01100 Viterbo, Italy
[j] Department of Biosystems and Technology, Swedish University of Agricultural Sciences, Box 190, SE-234 22 Lomma, Sweden
[k] Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia, Italy

This paper is in preparation for Animal.

Highlights

- There is a growing interest in the integration of animal welfare within Life Cycle Assessments (LCA) of animal-based food production systems.
- The review discusses the performance of 11 available approaches to account for animal welfare in LCA against a set of previously identified criteria.
- Results show high scores on the ability of the method to accurately estimate animal welfare were found to be associated with low applicability.
- Many approaches only consider one domain of animal welfare, simplifying the assessment, but limiting its accuracy, and often there is only a limited connection to a functional unit.
- Building on social LCA approaches while further developing the connection of indicators with the functional unit is key to increase both accuracy and applicability of methods for standard LCA.

Abstract

Life cycle assessment (LCA) represents an invaluable method for quantifying the sustainability trade-offs of different livestock systems, however animal welfare is seldom integrated within such assessments.

This review focused on studies that integrated animal welfare and life cycle assessment (LCA), selecting only peer-reviewed research related to livestock farming published in English after 2012. Eleven methods were evaluated based on a set of established general LCA criteria: credibility, transparency and reproducibility, fairness and acceptance, robustness, and applicability. In addition, specific criteria for incorporating animal welfare into LCA were applied, including accuracy, which reflects the ability to assess welfare across diverse production systems, and coherence, which refers to relevance across all stages of an animal's life.

The study found very few methods that integrate animal welfare assessments with LCA, with methodological complexity and data collection forming key barriers. Most standard LCAs integrating animal welfare focussed on few and easily attainable indicators with a limited connection to the functional unit, which limited their accuracy and prevented adequate coverage of the complexity of animal welfare. Social LCAs tended to perform better due to increased numbers of indicators covering wider animal welfare topics. Utilising approaches from social LCAs while ensuring the functional unit is linked to all indicators could allow standard LCA to accurately integrate animal welfare.

Keywords: Life Cycle Assessment, Livestock, Animal Welfare

Introduction

Life Cycle Assessment (LCA) is widely used to evaluate the impact of crop and livestock systems and related products on a range of environmental categories such as climate change and eutrophication (e.g., Flysjö et al., 2012; Kalhor et al., 2016; Grossi et al., 2018; Poore and Nemecek, 2018). It provides a quantitative analysis of environmental impacts across a product's entire life cycle, facilitating comparative evaluations and highlighting areas for improvement, ultimately fostering innovation, and policy development for enhanced sustainability (Notarnicola et al., 2017). Furthermore, the application of publicly available standardised methodologies in LCA enables a degree of transparency and consistency (ISO, 2006).

Despite the widespread adoption of environmental, social and other LCAs across the agri-food industry, LCAs of livestock systems often lack the level of detail required to enable real life decision making and do not consider wider food system aspects such as animal welfare (Sonesson et al., 2016). In particular, product-based LCAs tend to focus on the production function of systems rather than wider societal and environmental outcomes, e.g. regarding biodiversity or animal welfare (van der Werf et al., 2020). LCA studies on livestock furthermore underline the need to improve methods for capturing carbon sequestration, particularly through grassland management; crop-livestock interactions; circular economy aspects; impacts on biodiversity; food-feed competition and nutritional aspects (Goglio et al., 2015; Grossi et al., 2018; Kramer et al., 2018; Sonesson et al., 2019; van der Werf et al., 2020).

In order to further develop LCAs of livestock systems, a participatory approach identified key topics for method review and development (Goglio et al., 2023). These included: 1) Food, feed, fuel and biomaterial competition, crop-livestock interaction, circular economy; 2) Biodiversity; 3) Animal welfare; 4) Nutritional Aspects; and 5) GHG emission issues. The general criteria identified to evaluate existing LCA methods against were: 1) Credibility; 2) Transparency and Reproducibility; 3) Fairness and acceptance; 4) Robustness; and 5) Applicability (Goglio et al., 2023).

Animal welfare is of great and growing concern to European consumers (European Commission, 2022; European Parliament, 2023) and the importance of animal welfare in relation to sustainability has been highlighted, e.g. by Keeling et al. (2019). Hence, animal welfare needs to be integrated into holistic sustainability assessments of animal-based food production. Integrating animal welfare in a sustainability assessment such as LCA would allow for a more comprehensive understanding of production impacts and significantly improve the assessment, support better decision making by producers, policy makers and consumers and improve the sustainability of food systems overall (Hellweg and Milà i Canals, 2014; Fan et al., 2015; Lanzoni et al., 2023).

The World Organization for Animal Health defines animal welfare as "the physical and mental state of an animal in relation to the conditions in which it lives and dies" (OIE, 2019), which links to the three social concerns about animal welfare formulated by Fraser et al. (1997): positive health and functioning of animals, positive affective states (e.g., absence of prolonged fear or pain and experience of pleasure), and their ability to live a natural life by using natural adaptations and capabilities. The *Five Domains* approach understands animal welfare as combinations of conditions in different domains, which can be assessed as both negative and positive (Mellor and Reid, 1994). It includes the domains of nutrition, environment, health, behaviours, mental state.

The European *Welfare Quality®* project described a variety of indicators to assess animal welfare of livestock species on farm and at slaughter (Blokhuis et al., 2019; Welfare Quality, 2009 a, b, c). They are grouped under the following four principles: good feeding, good housing, good health and appropriate behaviour (Botreau et al., 2007; Blokhuis et al., 2019). However, resource- (e.g. housing) or management-based measures (e.g. feeding strategies) provide only partial information about the animals' welfare in particular situations. Animal- or outcome-based measures reflect the actual welfare state of the animals in terms of their behaviour, fearfulness, health, physical condition, etc. (Blokhuis et al., 2010). Therefore, it is necessary to include mental states as a domain when reviewing methods that aim at including animal welfare in LCA. For this reason, it was decided to use the *Five Domains* approach as a framework in this method review.

There is no consensus amongst LCA practitioners on the best existing practice for the incorporation of animal welfare in LCA, in a way that balances the quality of the measurement and practical considerations. Lanzoni et al. (2023) and Turner et al. (2023) worked on this topic in their scoping review of animal welfare integration in LCA studies, yet questions remain both on the criteria relevant to evaluate LCAs in general, and animal welfare in particular. This paper addresses this gap by evaluating existing methods against predefined quality criteria from both a general LCA perspective and an animal welfare-specific perspective, assigning scores to compare their performance. This helps to identify areas of successful integration as well as remaining issues and thus directs further method development.

Methods

This paper builds on work by Lanzoni et al. (2023) who reviewed the challenges of integrating animal welfare indicators into LCA, by using an additional inclusion criteria (publication integrates animal welfare measurements into LCA or proposes a method to do so) and by providing a specific assessment score for the performance of the methods reviewed (Table 5). The review is preceded by a harmonisation process which streamlined criteria for reviewing LCA methods considering both general LCA requirements and aspects linked specifically to the five key areas of livestock LCA development introduced in the introduction (Goglio et al. 2023). The identified criteria were used to assess LCA methods applied to livestock systems, as suggested in previous research for social LCA (S-LCA) (Macombe et al., 2018). The general criteria were applied to review methods for all the identified key areas for livestock LCA development, while the specific criteria differed depending on which key area was assessed. In this review, the specific criteria were selected to review methods combining animal welfare assessment and LCA.

REVIEW CRITERIA SELECTION

The criteria selection included expert workshops and surveys to establish both current focus areas for methodological development and review criteria as described in Goglio et al. (2023). The general criteria are summarised in Table 1.

In addition to the general criteria, specific criteria to evaluate LCA methodologies including animal welfare assessments were defined using a combination of expert knowledge and literature review, involving a working group of 3-4 individuals. *Table 5.* outlines the criteria specific to the key areas of animal welfare applied in the method review with their different evaluation levels. A description of the specific criteria is provided below.

Accuracy

The accuracy for animal welfare corresponds to the capacity of the assessment method to approximate the degree of animal welfare in different production systems. For this reason, it is expected that the method considers the following animal welfare aspects based on the *Five Domains* model: nutrition, environment, health, behaviours, mental state. Measurements of these criteria may be animal based, resource based, or management based. Level 1 is assigned to LCA methodologies that account for only one of the above animal welfare domains. Level 2 is assigned to LCA methodologies which account for two animal welfare domains. Finally, Level 3 is assigned to LCA methodologies which account for more than two animal welfare domains (*Table 5.*).

Coherence across the livestock value chain

Coherence across the livestock value chain is a specific criterion which describes the ability of the LCA methods to be used for different stages in the animal's life. Level 1 is assigned to LCA methodologies which only consider any welfare during the transport and slaughter phases. Level 2 is assigned to LCA methodologies which can only be applied at the farm phase. Level 3 is assigned to LCA methodologies which can be applied to all phases of the livestock value chain (*Table 5.*).

METHOD REVIEW

After the definition of the review criteria, searches were performed in Scopus, Web of Science and Google Scholar to find relevant, peer-reviewed papers integrating animal welfare assessment and life

cycle assessment. The chosen timeframe excluded studies which were published before 2012. The temporal boundary was chosen to coincide with the creation of the UN Sustainable Development Goals (SDGs) at the Rio+20 summit (Sachs, 2012).The search included the following terms:
"Assessment", "measure*", "protocol", "level*", "animal welfare", "animal well-being", "farm*", "transport", "slaughter*", "animal", "resource", "management", "livestock", "pig*", "cattle", "dairy", "sheep", "goat", "poultry", "layer*", "laying hen*", "broiler", "chicken*", Scientific names of the species and different LCA search terms ("Life Cycle Assessment", "LCA", "Life Cycle Analysis").
The searches on Web of Science, Scopus and Google Scholar initially resulted in 12,768 papers, reports and articles. In a first screening round of titles we identified 158 papers as potentially relevant for our review. Only research and discussion papers as well as reviews were included, but book chapters, conference proceedings and grey literature were excluded. The selection criteria applied are listed in **Error! Reference source not found.**. Results were reduced to 93 papers by removing duplicates. The remaining papers went through a second screening of abstracts, and in cases where specific inclusion and exclusion criteria was not clear in the abstract full text screening was required, to only include papers that either describe the performance of an LCA that includes animal welfare indicators or propose a methodology to do so. After the second screening round 22 papers remained. One study was included from outside of the timeframe due to the limited number of available publications combining LCA and animal welfare assessment (Müller-Lindenlauf et al., 2010). A full-text screening of the 22 papers led to the exclusion of further 12 papers as they did not meet the inclusion criteria outlined in **Error! Reference source not found.**. As the review progressed, another paper was included, which was published at a later point than the original paper search (Turner et al., 2023), so that a final number of 11 papers was included in the current review (Table 3). These papers were scored for the general and specific criteria as described above. Data analysis involved the calculation of means for each criterion and method as well as calculating the distribution of scores for each criterion.

Table 3 Eligibility Criteria for publications integrating animal welfare in LCA

Inclusion	Exclusion
Publication related to livestock farming	Publication not related to livestock farming; Publication focused on fish farming
Articles published in peer-reviewed scientific journals	Publications that have not undergone a peer-review process (e.g. book chapters, conference proceedings, reports)
Publication focused on Life Cycle Analysis, including Social LCA	Publication not focused on Life Cycle Analysis, including Social LCA
Publication integrates animal welfare measurements into LCA or proposes a method to do so	Publication does not consider animal welfare
Published in 2012 or later	Published before 2012
Published in English	Published in a language other than English

The searches on Web of Science, Scopus and Google Scholar initially resulted in 12,768 papers, reports and articles. In a first screening round of titles, we identified 158 papers as potentially relevant for our review as we included only research and discussion papers as well as reviews but excluded book chapters and conference proceedings or grey literature. The selection criteria applied are listed in **Error! Reference source not found.**. Results were reduced to 93 papers by removing duplicates. The remaining papers went through a second screening of abstracts, and in some cases full text item, to only include papers that either describe the performance of an LCA that includes animal welfare indicators or by proposing a methodology to do so. After the second screening round 22 papers remained. We also included one study outside of the timeframe due to the limited number of available publications combining LCA and animal welfare assessment (Müller-Lindenlauf et al., 2010). A full-text screening of the 22 papers led to the exclusion of further 12 papers as they did not meet the inclusion criteria outlined in Table 3. As the review progressed, another paper was included, which was published at a later point than the original paper search (Turner et al., 2023), so that a final number of 11 papers was included in the current review (**Error! Reference source not found.**). These papers were scored for the general and specific criteria as described above. Data analysis involved the calculation of means for each criterion and method as well as calculating the distribution of scores for each criterion.

RESULTS

Table 3 provides an overview of the assessed papers and the welfare indicators used by the different methods, which are described in detail for each paper below. Overall, there is some variety in the methods applied and indicators used. Four of the reviewed papers cover only one domain of animal welfare, three cover two domains and two cover more than two domains (**Error! Reference source not found.**).

Laying hens and broilers are the livestock species covered by the largest number of papers in this review, followed by cattle (*Figure 5 Number of papers covering different animal species*). Only one paper assessed sheep welfare. Results are presented based on the livestock types covered. Among the five domains of animal welfare, aspects related to animal health (e.g., mortality, foot lesions) were the most frequently assessed, followed by environmental aspects (e.g., stocking density, outdoor access). In contrast, aspects related to nutrition (e.g., naturalness, fibre content, pasture access), behaviour (e.g., tail biting, rooting), and emotional state (e.g., absence of fear) were assessed less frequently (*Figure 6*).

DESCRIPTION OF ASSESSED KEY METHODOLOGIES

POULTRY

Five of the reviewed studies focused on animal welfare in poultry farming. Boggia et al. (2019) performed an LCA and Life Cycle Costing (LCC) to test whether the installation of an innovative flooring system in broiler production impacts environmental, economic and animal welfare parameters and used a functional unit of 1kg of meat. Animal welfare data was collected daily for environmental (ammonia emissions) and health indicators (foot lesions and mortality). Specifically, ammonia emissions were calculated in ppm, while mortality and occurrence of foot legions were assessed as a percentage of the total flock. The same study collected data on the occurrence of foot lesions as an indicator for animal welfare on a daily basis, which is why the method scored low on applicability. Foot lesions at slaughter is, however, used in practice in e.g. Sweden, and works well as an economic incentive for farmers to ensure good quality litter in the broiler barns and thus avoid negative impacts, e.g., on weight gain (Berg, 1998) thereby improving welfare and profitability simultaneously. However, the method by Boggia et al. (2019) only applied to the farm phase, which explains the medium coherence score. Additionally, it considered only two domains of animal welfare (health and environment), so the score for accuracy is also medium. In addition, the animal welfare assessment was not integrated in the LCA, but was presented as a separate analysis (without reference to a functional unit) to show additional impacts of the innovative flooring system beyond those captured in the LCA and LCC.

Weeks et al. (2016) conducted a meta-analysis to model differences in mortality rates of laying hens and performed and LCA to quantify the environmental impacts of varying mortality levels with their functional unit being 1kg of collected eggs. They specifically assessed the health of aviary, barn, conventional cage, furnished cage, unfurnished cage, free range, and free range aviary systems by comparing the cumulative mortality percentage of laying hens in the different systems. The method scored low on accuracy due to it only relying on one indicator covering only one domain of animal welfare; and medium on coherence as it only applies to the farm phase. Mortality data being collected routinely potentially makes this method easier to apply. However, as the animal welfare assessment was not integrated into the LCA, but rather the LCA was performed for modelled mortality levels to see how those impacted environmental externalities, the method scored high for applicability.

Leinonen et al. (2014) performed an LCA to determine the effects of animal welfare enhancing changes to the impacts of broiler production systems (i.e. lower stocking density of an indoor system and combining this with heat exchanger for ventilation), with their functional unit being 1kg of expected carcass weight. They specifically assessed the environmental indicator initial stocking density of indoor barns (birds/m^2). They compared systems with standard indoor housing, low density housing, as well as low density housing with heat exchangers. This study scored low on the accuracy criterion as it only covered one domain of animal welfare (i.e. environment), and medium on coherence as it only applied to the farm phase. Applicability was scored as high, as the stocking density can be assessed easily on farms. Yet, similar to Weeks et al. (2016), the focus was on changes to the production system and how this impacts environmental externalities and animal welfare, rather than an assessment linked to the product's functional unit.

Turner et al. (2023) propose a S-LCA method adopting a reference-scale approach and apply it for the assessment of laying hens in Canada. They identified the area of protection, stakeholders, impact categories and subcategories, inventory indicators and data requirements, and characterization factors necessary for their assessment, based on a review of literature on best practice in animal welfare science and LCA. They tested their method with a case study assessing trade-offs of different housing systems in the Canadian egg industry, using 1 tonne of eggs as the functional unit. They specifically included environmental indicators (e.g. stocking density, prevision of nests and perches), health (e.g. mortality rates and foot condition), behaviour (e.g. presence of injurious behaviours) and mental state (e.g. excessive nervousness). The study scored each indicator on 0-1 scale, with a higher score indicating better animal welfare. The study scored high on robustness, as methodological components such as impact categories and data requirements were based on best practices in animal welfare and LCA. While the authors state that the method can be transferred to other species, it received a medium score for fairness and acceptance, as this would require some additional development. The method applied scored high on accuracy, since it covered all but the nutrition domain. The method also followed an approach of selecting animal welfare indicators related to animals' biological function, natural behaviour and affective state based on the concept by Fraser et al. (1997). This approach includes 19 indicators across four of the five animal welfare domains, except nutrition, which led to a high score for accuracy. The method scored medium on applicability due to the need to collect observation based on-farm data; and it scored medium on coherence across the supply chain as it only covered the farm phase.

Tallentire et al. (2019) propose a framework to incorporate animal welfare assessments of chickens into S-LCAs, calculating the overall animal welfare risk that chickens were exposed to using weighted sums after the risk for each indicator was determined using the Social Hotspots Database methodology. These calculations considered the environmental indicator of stocking density, as well as health indicators of mortality, and carcass condemnation rate to compare farming practices in several countries in Europe, using a functional unit of 1kg of chicken meat. Each indicator was scored 0-1, however, unlike Turner et al. (2023), a lower score indicated better animal welfare. This method received a medium score for the specific criteria accuracy due to including two domains of animal welfare with their indicators, i.e. environment (stocking density) and health (mortality, dead on arrival, carcass condemnation); and high for coherence across the value chain due to being applicable to farm (housing, mortality), transport (dead on arrival) and slaughter phases (carcass condemnation). However, the last indicator can be indicative of low animal welfare states during different phases, so that the distinction of when a condemnation happened might not be separable in this indicator. As the indicators used are routinely collected, at least for chickens (Tallentire et al., 2019), the method also scored high for applicability. Carcass condemnation is recorded for all species at slaughterhouses routinely, making the indicator transferable to other species, and mortality, stocking density and dead-on arrival are indicators that can also be used in other species, which is why the method received a medium score for Fairness and Acceptance overall.

CATTLE

Two studies specifically looked at animal welfare as related to cattle farming. Müller-Lindenlauf et al. (2010) assessed 27 organic dairy farms for their environmental impact using LCA, milk quality and animal welfare assessments using a scoring system, that enabled the integration of animal welfare with the LCA results in an overall index. This included indicators covering three of the animal welfare domains described above, namely environment (stocking rate, hours of pasturing and whether the system was free range or not), nutrition (e.g., fibre intake) and health (somatic cell count, as well as horn amputation). Environmental categories were scored out of 30, while nutritional and health categories were both scored out of 15 points each, with lower scores corresponding to better animal welfare performance. The overall animal welfare score was calculated on a scale of 0-10 alongside other impact categories with various functional units (i.e. climate impact g CO_2-equiv./kg milk, or land demand ha/1000 kg milk). All indicator scores were standardised and combined to calculate an overall sustainability index on a 0–10 point scale (with lower scores corresponding to a better sustainability performance). As the study included three animal welfare domains, it scored high for accuracy. As these indicators only evaluate the situation on farm and exclude the transport and slaughter phases, the coherence score was medium. The study scored medium on applicability, as information needs to be collected from the individual farm, although the data could be readily available on many commercial farms.

Zucali et al. (2016) introduced a scoring system for assessing animal welfare on dairy farms, which they combined with an LCA and lab analysis of milk's nutritional, microbial and nutraceutical status with a functional unit of 1 kg of fat and protein corrected milk. Data was collected on 29 farms directly and the animal welfare scoring included a selection of health and feeding indicators of the Welfare Quality® (2009) assessment for cattle, namely Body Condition Score, absence of lameness, absence of diarrhoea and absence of claw overgrowth. Due to this data having to be collected on farms, the method scored low on applicability. Likewise, a low score was obtained for coherence, as it does not apply to the transport and slaughter phases. The score for accuracy was medium, due to the scoring system considering indicators covering the animal welfare domain "health" and "feeding". As in other studies, the animal welfare assessment was not included in the LCA, but it was performed separately, and results were scored alongside other categories (e.g. milk's nutritional profile) for a holistic assessment.

PIGS

Of the reviewed studies only Zira et al. (2020) focussed specifically on pig farming. They performed a S-LCA to evaluate the risks of negative social impacts (e.g., low wages, deforestation, disease prevalence, animal welfare impacts) linked with the production of conventional and organic pork in Sweden, using a functional unit of 1000kg pork (fork weight). Including pigs as a stakeholder group, authors calculated Social Risk and used an Analytical Hierarchical Process to determine the "Social Risk Time", indicating the level of exposure of different stakeholder groups to different social risks, as well as the "Social Hotspot Index", which "indicates the risk of negative social impacts relative to the worst case scenario for a given stakeholder and/or subsystem" (Zira et al. 2020: 1970). Specifically, they included indicators for the environment (e.g. percentage of pigs with access to daylight and slatted floor, nutrition (percentage of pigs provided roughage as feed), health (e.g. injuries per pig and prevalence of shoulder legions), as well as behaviour and mental state (e.g. percentage of pigs with bitten tails). Data for the farm and slaughter phases were collected from articles, reports, websites, interviews and survey data. The method only reaches a medium score for applicability, as various impact categories, which fed into one aggregated score, had to be weighed by experts to determine the social risk for the stakeholder group “pigs”. Yet, a high score was reached for coherence as well as accuracy, as the method presents indicators for both the farm and slaughter phase and used indicators covering all of the above described domains of animal welfare (i.e. health, environment, nutrition, behaviour and emotional state).

SHEEP

The only reviewed study to focus on sheep farming was Geß et al. (2020). They add an animal welfare indicator to an LCA comparing lamb production in semi-intensive and semi-extensive systems by measuring cortisol accumulation in the wool to assess their levels of chronic stress. Wool samples were taken at 30-day intervals over the course of 4 months to provide an indicator on mental state, with 1kg of lamb meat being used as the functional unit for the LCA.

Since data collection takes a lot of effort, the method scored low on applicability. This indicator depicts long-term stress levels (Stubsjøen et al., 2015; Fürtbauer et al., 2019) if measured at regular intervals and thus has the potential to score high on the coherence category. Yet, short-term experiences close to the animals death (transport, pre-slaughter and slaughter experiences) need to be assessed with accompanying measurements, e.g. blood measurements of cortisol, lactate and glucose (Petherick et al., 2009; Edwards et al., 2010; Broom, 2011; Hultgren et al., 2022) or behavioural indicators (Wilhelmsson et al., 2023). Accuracy was scored low, despite the relationship of low chronic stress to an absence of (multiple) aspects of animal welfare. The method is not integrated into the LCA, but rather an additional assessment to measure animal welfare alongside environmental impacts.

CATTLE, POULTRY, PIGS AND OTHERS

Two studies looked at animal welfare across a wide range of livestock types including cattle, poultry, pigs, aquaculture and insects. In their paper, Paris et al. (2022) performed an LCA assessing male and female diets in a German state against sustainable diets under the One Health approach by including additional indicators to measure diet related human health outcomes and animal welfare. Animal welfare was added as an additional impact category including various environmental indicators (e.g. number of animals effected, quality of life, life duration and slaughter duration) and using a functional unit of 4.1kg and 3.6kg per capita per person for men and women respectively. It was assessed according to the method proposed by Scherer et al. (2018) and is thus expressed as *animal life years suffered*, *loss of animal lives* and *loss of morally-adjusted animal lives.*

Scherer et al. (2018) propose a framework for including animal welfare assessments in LCA by calculating the impacts on *animal life years suffered (ALYS)*, *loss of animal lives (AL)* and *loss of morally-adjusted animal lives (MAL)*. All three indicators consider the lifetime of an animal as well as the number of animals affected by conditions of the production system and take into account the farm, transport to slaughter and slaughter stage. The indicator is linked to a functional unit of 1 kg of meat. The calculation of ALYS assumes a suffering state, from which death means salvation. Animal welfare impacts are accounted as the loss of animal welfare, expressed as the number of years an animal has to live in the state of suffering The calculation of AL is based on the assumption that ultimately animals strive for survival, and thus the life lost needs to be considered as well as the quality of life when assessing animal welfare. The third indicator MAL values animal lives differently, depending on their self-awareness and sense of time. A moral value is allocated to the animals due to their expected intelligence relative to humans, estimated by the number of cortical or total neurons, or brain mass. The approach taken is innovative and beneficial in that it considers the number of individuals affected to produce a kg of product. Paris et al. (2022) have demonstrated that the method is feasibly integrable into LCAs of food products. Nevertheless, it could be argued that the number and nature of ethical assumptions that are made negatively affect the robustness of the method. Animal welfare assessments should first focus on measuring hazards for and consequences of different setups for animal wellbeing. The question of what level of animal welfare is ethically acceptable comes after any scientific assessment and should not be mixed up with it beforehand. The robustness is further compromised by uncertainties regarding the MAL indicator of a) using intelligence as a proxy for self-awareness when it is indeed not a measure for it (Scherer et al., 2018) and b) using individual indicators such as the number of neurons, cortical neurons or brain mass to approximate intelligence, when research suggests that it is rather a combination of different factors that determine intelligence (Dicke and Roth, 2016). The method scored high on the criterion for *coherence across the value chain* due to including the transport to, pre- and slaughter phases in the calculations of ALYS and AL, however it needs to be stressed that this does only account for the amount of time that animals suffer during the slaughtering process and does not consider other factors related to the level of suffering (e.g. whether stunning is applied, Scherer et al., 2018).

SCORING OF METHODS

For the general criteria, results were mixed but overall satisfactory. Methods scored very high on "Robustness" with an average score of 2.8 (Figure 3, Table 4) out of a maximum score of 3, and 80% of papers reaching the highest score (Figure 4). Scores were also very good for "Completeness" with an average of 3.7 out of a maximum score of 4, with 70% of the papers reaching the highest score; and

"Transparency and Reproducibility", which had an average score of 2.9 out of 3, and all but one method in the highest category.

A low score was reached for the category "Fairness and Acceptance" with an average of 1.7 out of 3, and 90% reaching only a low or medium score. This can be explained with some methods being only suitable for use on certain species or products. Another low score of 1.9 was reached for the category "Applicability" (Table 4), which reflects the different levels of complexity in collecting the selected animal welfare indicators, e.g. from data which need to be collected on farms, versus data routinely collected in slaughterhouses (Figure 4).

For the specific criteria, the majority of papers reached only low scores (Figure 3). For "coherence across the value chain", only 33% of methods (n=3) were applicable to all phases of the value creation process, while 67% of methods could either only be used for the farm phase or included slaughter but no indicators to account for transport to the slaughterhouse. For the "Accuracy" criterion, 44% of papers received a low score, meaning that only one domain was used to determine the animal welfare status, while 30% (n=3) covered two and 30% (n=3) more than two (Figure 4).

The highest ratings are reached by the paper by Tallentire et al. (2019) with an average score of 2.9 due to very good performance across most of the assessed criteria (Table 4). The study by Zira et al. (2020) reaches an overall score of 2.7 with only two of the general criteria reaching high scores, the same study scores high on coherence and accuracy, due to the inclusion of an extensive number of indicators covering all five animal welfare domains. The study by Turner et al. (2023) also reached an overall score of 2.7, due to high scores across all categories except coherence across the value chain and applicability.

The approach used by Paris et al.(2022), using the Scherer et al. (2018) method achieves a score of 2.6 as it scores high to very high for three of the general criteria and one of the specific criteria. Yet, it reached only the lowest score for accuracy as it only includes one indicator to assess quality of life (Scherer et al., 2018). The method scored high for coherence as indicators calculate the whole life to slaughter and include slaughter and transport time. The medium applicability score is related to data availability for measuring *Life Quality* (e.g. stocking density, access to pasture), that need to be collected from farms. Data collection could become more comprehensive if more domains were to be covered by including more indicators.

Leinonen et al. (2014) reach a score of 2.6 scoring high or very high for the general criteria except fairness and acceptance but reaching low and medium scores on the specific criteria. Geß et al. (2020) received the lowest score of 2.0 (Table 4).

Discussion

As LCA practitioners, methods are being sought that allow us to integrate data on animal welfare into the assessment in a way that balances effort and robustness. In an ideal world, this requires indicators of which data a) can be related to a functional unit, b) is continuous, c) is relatable to all stages of an animal's life, d) is readily available or easy to collect, and e) is a satisfactory representation of the conditions the animal has lived in.

Before undertaking the review, the expectation was to see a clear trade-off between applicability and accuracy, as previously discussed for soil C in LCA of agriculture (Goglio et al., 2015), meaning that high scores for accuracy would be linked to low scores for applicability. While this is not reflected by all scores, it can be confirmed that there is a tendency towards this expected outcome. The three methods scoring high in accuracy received low or medium scores for applicability, through requiring either the collection of animal-based data (Müller-Lindenlauf et al., 2010; Turner et al., 2023), or expert modulation of results (Zira et al., 2020). However, Tallentire et al. (2019) reaches a high applicability and medium accuracy score as it only includes indicators covering animal health and environment, which are easier to collect than other domains such as behaviour, emotional state or nutrition.

This relative ease of data collection is generally reflected by health and environment domains being covered the most. Health data it is often routinely recorded, and environment data (e.g. stocking rate) is, in theory, also easy to obtain from farms, as discussed previously in LCA research (Miller et al., 2006). This is in line with the review by Lanzoni et al. (2023), which found the environment domain (Mellor and Reid, 1994; Mellor et al., 2020) to be covered the most, followed by health, while mental state was included the least. Many of the assessed papers only consider one domain of animal welfare, which further simplifies the assessment as different types of data and scores do not need to be weighted. This result differs from the review undertaken by Lanzoni et al. (2023), which found that 29% of studies only considered one domain of animal welfare, 25% two domains, 21% three domains and 13% four domains. This difference is likely explained by the greater number of studies included in their review, compared to this discussion paper.

While considering one indicator that only covers a single domain of animal welfare, as it is the case for four of the reviewed methods, limits the accuracy, adding more animal welfare indicators to the LCA makes the analysis more complex. This is reflected by roughly half of the methods (n=5), which either added animal welfare assessments as a supplement to the LCA, rather than truly integrating it (Zucali et al., 2016; Geß et al., 2020), or modelled the impacts of welfare-related changes (new housing system and decreased mortality) on environmental indicators (Leinonen et al., 2014; Weeks et al., 2016; Boggia et al., 2019). However, two studies do integrate animal welfare as an impact category into LCA (Müller-Lindenlauf et al., 2010; Scherer et al., 2018); and three into S-LCA (Tallentire et al., 2019; Turner et al., 2023; Zira et al., 2020). Integrating animal welfare into LCA thus needs to balance the complexity of analysis with both the meaningfulness of animal welfare indicators and the feasibility of collecting them. This balancing is reflected in the scoring framework used in this review, which is limited by not differentiating between how good or capable indicators are for approximating animal welfare, as it only counts how many domains are covered. Yet, a low scoring single indicator may well be a good predictor of animal welfare (e.g. cortisol in wool).

The highest scoring methods in this review are S-LCAs (Tallentire et al., 2019; Turner et al., 2023; Zira et al., 2020) and thus include the collection of comprehensive sets of social indicators, including animal welfare. These methods are the only ones scoring medium or high across the specific criteria and applicability. Even if it is not feasible to integrate the same amount of data as collected for a S-LCA into a standard LCA, there are important aspects that can be included, e.g. on indicator selection or weighting of results or scores. Most of the reviewed papers utilise management or resource based indicators, sometimes alone, sometimes in combination with animal-based measures, such as stocking density (Leinonen et al., 2014; Tallentire et al., 2019), access to daylight (e.g., Zira et al., 2020) or pasture(e.g., Müller-Lindenlauf et al., 2010; Scherer et al., 2018), and the existence of nesting boxes and perches (Turner et al., 2023). Animal-based measurements are used by two S-LCA papers and three standard LCAs in this review, including the occurrence of lesions (Boggia et al., 2019; Turner et al., 2023; Zira et al., 2020), body condition score (e.g., Zucali et al., 2016), milk cell count (Müller-Lindenlauf et al., 2010) or cortisol in wool (Geß et al., 2020). This shows that despite S-LCAs typically covering a wider range of animal welfare indicators, it is not impossible to obtain and use animal-based indicators measuring consequences of environmental and management conditions in standard LCA.

It is recognized that determining the meaningfulness of animal welfare indicators is subject to a widespread scientific debate, (Miele and Evans, 2010; Broom, 2011; Mellor et al., 2020; Stokes et al., 2022) which is beyond the scope of this paper. Proposing indicators that can be feasibly included in LCA will require a wider scientific consensus as expressed through the development of animal welfare assessment frameworks. Nevertheless, it can be stressed that most of the methods reviewed in this discussion paper are somewhat lacking in their ability to adequately approximate the conditions an animal has lived and died in. While the S-LCA methods proposed by Turner et al. (2023) and Zira et al. (2020) can be considered partially exempt from this due to their coverage of animal welfare domains, most of the methods discussed here do not consider these post-farm phases, which is a significant drawback to evaluating animal welfare along the value chain.

Despite the accuracy of methods assessed, one of the key aspects from an LCA perspective is the question on whether these animal welfare indicators can be related to the same functional unit as other impact categories in the assessment. A few methods presented here have done this successfully (Scherer et al., 2018; Tallentire et al., 2019; Turner et al., 2023; Zira et al., 2020). As this involves a weighting and an indicator selection process, applying these approaches in future research is necessary to increase robustness and make it applicable for different livestock species (Lanzoni et al., 2023).

Scherer et al. (2018) and Müller-Lindenlauf et al. (2010) present methods that integrate animal welfare into standard LCAs as an additional impact category, but only the former method relates to the same functional unit as the remaining analysis. Müller-Lindenlauf et al. (2010) use a ranking system for the animal welfare assessment, which requires weighting and is not linked to the functional unit. The method of Scherer et al. (2018) is the only approach that captures animal welfare referred to a functional unit of a standard LCA. Yet, the method has two major drawbacks: 1) the *Life Quality* criterion relies on only a single animal welfare indicator covering only one domain (e.g. environment); and 2) the strong ethical stances that are imposed by the different indicators, offering different views and values of animal lives in production systems.

Conclusions

The number of methods that combine or integrate animal welfare assessments with LCA is very small, with very few studies involving sheep, or pigs. The available studies which were reviewed differ in the number of animal welfare domains covered, the phases of an animal's life that they consider and how readily data is available. It was found that animal welfare integration in standard LCAs is lacking in accuracy with regard to a meaningful assessment of the lives of animals in farming systems. This is different for S- LCAs, which are the highest scoring methods in this review, due to the wider range of indicators they can include, thus often covering many animal welfare domains.

From an LCA perspective, the complexity of analysis (i.e. a methodology of integrating animal welfare into a standard LCA) and the feasibility of data collection are the biggest challenges in relation to the assessed methods. The only method which relates the animal welfare score to the same functional unit as standard LCA also requires an ethical stance on the value of animal lives in production systems, which makes it less robust, and the same approach only considers one indicator to approximate animal welfare.

Going forward, combining methods that can relate animal welfare to a functional unit and at the same time offer a satisfactory level of accuracy for different types of livestock species is important. It is recognised that this may come at the trade-off of decreased applicability and more complex data collection, but it is essential for developing a method that can produce robust and accurate results. Data available through real-time video monitoring and AI may be a potential avenue for widening the realm of available information for at least some husbandry systems in the future. Moreover, metrics and indicators that have been used for measuring and assessing animal welfare outside of LCA could be reviewed alongside an exploration of how these could be integrated into the LCA method. This could help further explore additional aspects of animal welfare, how it can be measured, and the potential for different metrics to complement LCA alongside any challenges they may pose for integration into LCA. While we did not consider the meaningfulness of animal welfare indicators (i.e. causal links between indicators and well-being) in our review, we consider it to be a crucial aspect of including animal welfare in LCA and would thus stress the need to consider debate around this topic in any method application and development.

Funding

This research was carried out within the "Pathways for transitions to sustainability in livestock husbandry and food systems (PATHWAYS)" project, part of the European Union's Horizon 2020 Research and Innovation Programme, under grant agreement No 101000395.

Declaration of competing interest

The authors declare that they have no conflict of interest.

References

Berg, L., 1998. Foot-pad dermatitis in broilers and turkeys: prevalence, risk factors and prevention. Department of Animal Environment and Health. Swedish University of Agricultural Sciences Acta Universitatis Agriculturae Sueciae. Veterinaria.

Blokhuis, H.J., Veissier, I., Miele, M., Jones, B., 2019. Safeguarding farm animal welfare. In: Vogt, M. (Ed.), Sustainability Certification Schemes in the Agricultural and Natural Resource Sectors: Outcomes for Society and the Environment. Routledge.

Blokhuis, H.J., Veissier, I., Miele, M. and Jones, R.B., 2010. The Welfare Quality® project and beyond: safeguarding farm animal well-being. Acta Agriculturae Scandinavica A, Animal Science, 60, 129-140.

Boggia, A., Paolotti, L., Antegiovanni, P., Fagioli, F.F., Rocchi, L., 2019. Managing ammonia emissions using no-litter flooring system for broilers: Environmental and economic analysis. Environmental Science & Policy 101, 331-340.

Botreau, R., Veissier, I., Butterworth, A., Bracke, M.B.M., Keeling, L.J., 2007. Definition of criteria for overall assessment of animal welfare. Animal Welfare 16, 225-228.

Broom, D.M., 2011. A history of animal welfare science. Acta Biotheor 59, 121-137.

Courboulay, V., Eugène, A., Delarue, E., 2009. Welfare assessment in 82 pig farms: effect of animal age and floor type on behaviour and injuries in fattening pigs. Animal Welfare 18, 515-521.

Dicke, U., Roth, G., 2016. Neuronal factors determining high intelligence. Philosophical Transactions of the Royal Society B: Biological Sciences 371, 20150180.

Edwards, L.N., Engle, T.E., Correa, J.A., Paradis, M.A., Grandin, T., Anderson, D.B., 2010. The relationship between exsanguination blood lactate concentration and carcass quality in slaughter pigs. Meat Science 85, 435-440.

European Commission, 2022. Animal welfare – revision of EU legislation - public consultation.

European Parliament, 2023. Animal welfare protection in the EU - strategy and law - briefing.

Fan, Y., Wu, R., Chen, J., Apul, D., 2015. A Review of Social Life Cycle Assessment Methodologies. In: Muthu, S.S. (Ed.), Social Life Cycle Assessment: An Insight. Springer Singapore, Singapore, pp. 1-23.

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S.F., 2012. The interaction between milk and beef production and emissions from land use change – critical considerations in life cycle assessment and carbon footprint studies of milk. Journal of Cleaner Production 28, 134-142.

Fraser, D., Milligan, B.N., Pajor, E.A., Weary, D.M., 1997. A Scientific Conception of Animal Welfare that Reflects Ethical Concerns. Animal Welfare 6, 187-205.

Fürtbauer, I., Solman, C., Fry, A., 2019. Sheep wool cortisol as a retrospective measure of long-term HPA axis activity and its links to body mass. Domestic Animal Endocrinology 68, 39-46.

Geß, A., Viola, I., Miretti, S., Macchi, E., Perona, G., Battaglini, L., Baratta, M., 2020. A New Approach to LCA Evaluation of Lamb Meat Production in Two Different Breeding Systems in Northern Italy. Frontiers in Veterinary Science 7.

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. Cleaner Production Letters 4, 100035.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. Journal of Cleaner Production 104, 23-39.

Grossi, G., Goglio, P., Vitali, A., Williams, A.G., 2018. Livestock and climate change: impact of livestock on climate and mitigation strategies. Animal Frontiers 9, 69-76.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109-1113.

Hultgren, J., Segerkvist, K.A., Berg, C., Karlsson, A.H., Öhgren, C., Algers, B., 2022. Preslaughter stress and beef quality in relation to slaughter transport of cattle. Livestock Science 264, 105073.

International Organization for Standardization, 2006. Environmental management — Life cycle assessment — Principles and framework (ISO Standard No 14040:2006).

Kalhor, T., Rajabipour, A., Akram, A., Sharifi, M., 2016. Environmental impact assessment of chicken meat production using life cycle assessment. Information Processing in Agriculture 3, 262-271.

Keeling, L., Tunón, H., Olmos Antillón, G., Berg, C., Jones, M., Stuardo, L., Swanson, J., Wallenbeck, A., Winckler, C., Blokhuis, H., 2019. Animal Welfare and the United Nations Sustainable Development Goals. Frontiers in Veterinary Science 6.

Kramer, G.F.H., Martinez, E.V., Espinoza-Orias, N.D., Cooper, K.A., Tyszler, M., Blonk, H., 2018. Comparing the Performance of Bread and Breakfast Cereals, Dairy, and Meat in Nutritionally Balanced and Sustainable Diets. Front Nutr 5, 51.

Lanzoni, L., Whatford, L., Atzori, A.S., Chincarini, M., Giammarco, M., Fusaro, I., Vignola, G., 2023. Review: The challenge to integrate animal welfare indicators into the Life Cycle Assessment. Animal 17, 100794.

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

Macombe, C., Loeillet, D., Gillet, C., 2018. Extended community of peers and robustness of social LCA. The International Journal of Life Cycle Assessment 23, 492-506.

Mellor, D.J., Beausoleil, N.J., Littlewood, K.E., McLean, A.N., McGreevy, P.D., Jones, B., Wilkins, C., 2020. The 2020 Five Domains Model: Including Human–Animal Interactions in Assessments of Animal Welfare. Animals 10, 1870.

Mellor, D.J., Reid, C., 1994. Concepts of animal well-being and predicting the impact of procedures on experimental animals.

Miele, M., Evans, A., 2010. When foods become animals: Ruminations on Ethics and Responsibility in Care-full practices of consumption. Ethics, Place & Environment 13, 171-190.

Mullender, S.M., Sandor, M., Pisanelli, A., Kozyra, J., Borek, R., Ghaley, B.B., Gliga, A., von Oppenkowski, M., Roesler, T., Salkanovic, E., Smith, J., Smith, L.G., 2020. A delphi-style approach for developing an integrated food/non-food system sustainability assessment tool. Environmental Impact Assessment Review 84, 106415.

Müller-Lindenlauf, M., Deittert, C., Köpke, U., 2010. Assessment of environmental effects, animal welfare and milk quality among organic dairy farms. Livestock Science 128, 140-148.

OIE, 2019. Terrestrial Animal Health Code. Chapter 7.1 Introduction to the recommendations for animal welfare.

Paris, J.M.G., Falkenberg, T., Nöthlings, U., Heinzel, C., Borgemeister, C., Escobar, N., 2022. Changing dietary patterns is necessary to improve the sustainability of Western diets from a One Health perspective. Sci Total Environ 811, 151437.

Petherick, J.C., Doogan, V.J., Venus, B.K., Holroyd, R.G., Olsson, P., 2009. Quality of handling and holding yard environment, and beef cattle temperament: 2. Consequences for stress and productivity. Applied Animal Behaviour Science 120, 28-38.

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science 360, 987-992.

Sachs, J.D., 2012. From millennium development goals to sustainable development goals. *The lancet*, *379*(9832), pp.2206-2211.

Scherer, L., Tomasik, B., Rueda, O., Pfister, S., 2018. Framework for integrating animal welfare into life cycle sustainability assessment. The International Journal of Life Cycle Assessment 23, 1476-1490.

Sonesson, U., Davis, J., Hallström, E., Woodhouse, A., 2019. Dietary-dependent nutrient quality indexes as a complementary functional unit in LCA: A feasible option? Journal of Cleaner Production 211, 620-627.

Sonesson, U.G., Lorentzon, K., Andersson, A., Barr, U.-K., Bertilsson, J., Borch, E., Brunius, C., Emanuelsson, M., Göransson, L., Gunnarsson, S., Hamberg, L., Hessle, A., Kumm, K.-I., Lundh, Å., Nielsen, T., Östergren, K., Salomon, E., Sindhöj, E., Stenberg, B., Stenberg, M., Sundberg, M., Wall, H., 2016. Paths to a sustainable food sector: integrated design and LCA of future food supply chains: the case of pork production in Sweden. The International Journal of Life Cycle Assessment 21, 664-676.

Stokes, J.E., Rowe, E., Mullan, S., Pritchard, J.C., Horler, R., Haskell, M.J., Dwyer, C.M., Main, D.C.J., 2022. A "Good Life" for Dairy Cattle: Developing and Piloting a Framework for Assessing Positive Welfare Opportunities Based on Scientific Evidence and Farmer Expertise. Animals (Basel) 12.

Stubsjøen, S.M., Bohlin, J., Dahl, E., Knappe-Poindecker, M., Fjeldaas, T., Lepschy, M., Palme, R., Langbein, J., Ropstad, E., 2015. Assessment of chronic stress in sheep (part I): The use of cortisol and cortisone in hair as non-invasive biological markers. Small Ruminant Research 132, 25-31.

Tallentire, C.W., Edwards, S.A., Van Limbergen, T., Kyriazakis, I., 2019. The challenge of incorporating animal welfare in a social life cycle assessment model of European chicken production. The International Journal of Life Cycle Assessment 24, 1093-1104.

Turner, I., Heidari, D., Widowski, T., Pelletier, N., 2023. Development of a life cycle impact assessment methodology for animal welfare with an application in the poultry industry. Sustainable Production and Consumption 40, 30-47.

van der Werf, H.M.G., Knudsen, M.T., Cederberg, C., 2020. Towards better representation of organic agriculture in life cycle assessment. Nature Sustainability 3, 419-425.

Weeks, C.A., Lambton, S.L., Williams, A.G., 2016. Implications for Welfare, Productivity and Sustainability of the Variation in Reported Levels of Mortality for Laying Hen Flocks Kept in Different Housing Systems: A Meta-Analysis of Ten Studies. PLoS One 11, e0146394.

Welfare Quality®, 2009. Welfare Quality® assessment protocol for cattle. Welfare Quality® Consortium, Lelystad, Netherlands.

Wilhelmsson, S., Andersson, M., Hemsworth, P.H., Yngvesson, J., Hultgren, J., 2023. Human-animal interactions during on-farm truck loading of finishing pigs for slaughter transport. Livestock Science 267, 105150.

Zira, S., Röös, E., Ivarsson, E., Hoffmann, R., Rydhmer, L., 2020. Social life cycle assessment of Swedish organic and conventional pork production. The International Journal of Life Cycle Assessment 25, 1957-1975.

Zucali, M., Battelli, G., Battini, M., Bava, L., Decimo, M., Mattiello, S., Povolo, M., Brasca, M., 2016. Multi-dimensional assessment and scoring system for dairy farms. Italian Journal of Animal Science 15, 492-503.

Tables

Table 4. Identified general criteria to assess LCA methods in the LCA of livestock systems (Goglio et al. 2023)

General criteria definition	Level 1	Level 2	Level 3	Level 4
Transparency and Reproducibility: Comprehensive documentation and mechanisms that allow reviewers to verify/review all data, calculations, and assumptions	LCA methodologies which do not allow reviewers to verify/review the results, calculations and assumptions.	LCA methodologies which could be reviewed together with the results, but some calculations and assumptions cannot be reviewed.	LCA methodologies which fully allows reviewers to verify/review the results, calculations and assumptions	
Completeness: quantification of the environmental impact including all material/energy flows and other environmental interventions as required for adherence to the defined system boundary, the data requirements, and the impact assessment methods employed	the quantification of the environmental impacts including all material/energy flows and other environmental interventions do not have adherence to the system boundary, the data requirements and the impact assessment methods employed	the quantification of the environmental impacts conforms either to the defined system boundary or the data requirements or the system method employed	the quantification of the environmental impacts conforms to two aspects between the defined system boundary, data requirements and impact assessment method employed	the quantification of the environmental impacts fully corresponds to the system boundary, data requirements and the impact assessment methods employed
Fairness and acceptance: associated with providing a level playing field across competing products, processes and industries. Exceptions must not relatively disfavour competitors. The role of interested parties and of review is strengthened for achieving broad stakeholder acceptance. Protecting confidential and proprietary information in confidential reports that are available exclusively to the critical reviewers.	the LCA methodology does not provide level playing field across products, processes and industries	the LCA methodology provides a level playing field for at least two products, processes and industries (e.g. beef and dairy; beef and pig)	LCA provides a level playing field for several products, processes and industries	
Robustness: associated in the RACER framework the following sub-criteria of providing a defensible theory, Sensitivity, Data quality, Reliability, Consistency, Comparability, Boundaries	the LCA methodology is not based on defensible theory, lacks sensitivity on certain environmental impacts either because of its reliability, comparability, the chosen system boundary or its comparability	the LCA methodology is based on a defensible theory but it lacks sensitivity, reliability, comparability and it is not in agreement with the system boundaries	the LCA methodology is based on a defensible theory with a satisfactory sensitivity, reliability, data quality, consistency, comparability and in agreement with the system boundaries	
Applicability: the ability of the method to be used by a wide range of LCA practitioners	the LCA method can be easily used with very limited LCA expertise and data availability	the LCA method can be used with either limited LCA expertise or data availability	the LCA method can only be used with LCA expertise and extensive data availability	

Table 5. Specific criteria definition and scale for animal welfare (Goglio et al., 2023; Supporting Table 5, adapted)

Specific criteria definition	Level 1	Level 2	Level 3

Accuracy: the capacity of the LCA method to capture the degree of animal welfare and the ability to make cause-effects relationships with the type of livestock production. The method should take into account the following animal welfare domains: nutrition, environment, health, behaviours, mental state.	the LCA method accounts for one animal welfare domain	the LCA method accounts for two animal welfare domains	the LCA method accounts for more than two animal welfare domains
Coherence across the livestock value chain: ability of the LCA method to be used along the livestock value chains for the relevant processes related to animal welfare (i.e. when the animal is alive).	the LCA method only covers transport and slaughter phases	the LCA method only covers the farm phase	the LCA method accounts for all phases of an animal's life (i.e. farm, transport, slaughter)

Table 6. Eligibility Criteria for publications integrating animal welfare in LCA

Inclusion	**Exclusion**
Publication related to livestock farming	Publication not related to livestock farming; publication focused on fish farming
Articles published in peer-reviewed scientific journals	Publications that have not undergone a peer-review process (e.g. book chapters, conference proceedings, reports)
Publication focused on Life Cycle Analysis, including social LCA	Publication not focused on Life Cycle Assessment, including social LCA
Publication integrates animal welfare measurements into LCA or proposes a method to do so	Publication does not consider animal welfare
Published in 2012 or later	Published before 2012
Published in English	Published in a language other than English

Table 7. Overview of methods and applied animal welfare hazards and indicators

Publication	Livestock type	Five Domains					Animal welfare hazards and consequences measured	
		Environment	Nutrition	Health	Behaviour	Mental state	Hazards measured (resource/management based)	Consequences measured (animal-based)
Boggia et al., 2019	Broilers	X		X			ammonia emissions	foot lesions, mortality
Leinonen et al., 2014	Broilers, laying hens	X					stocking density	
Geß et al., 2020	Sheep					X		cortisol levels in wool
Müller-Lindenlauf et al., 2010	Dairy cows	X	X	X			e.g., hours on pasture, access to free range, fibre content of ration, dehorning	Milk cell count
Paris et al., 2022	Cattle, pigs, laying hens and broilers, turkeys, fish, shrimp, honeybees						same as Scherer et al., 2018	
Scherer et al., 2018	Cattle, pigs, laying hens and broilers, salmon, shrimp, insects	X					days on pasture, stocking density	

Tallentire et al., 2019	Broilers	X		X			stocking density	mortality, dead on arrival, carcass condemnation
Turner et al., 2023	Laying hens	X		X	X	X	e.g., nests provided, perches provided, stocking density	e.g., mortality, feather & foot condition, injurious behaviour, excessive nervousness
Weeks et al., 2016	Laying hens			X				mortality
Zira et al., 2020	Pigs	X	X	X	X	X	e.g., access to daylight, slatted floor, space per pig, access to roughage, , access to water	e.g., occurrence of stress/fear, piglet mortality, lesions, tail biting, rooting behaviour
Zucali et al., 2016	Dairy cows		X	X				Body Condition Score, absence of lameness, absence of diarrhoea, absence of claw overgrowth

Table 8. Scoring of General and Specific criteria of reviewed papers

Publication	Transparency and Reproducibility	Completeness	Fairness and Acceptance	Robustness	Applicability	Accuracy	Coherence across the livestock value chain	Average score
Tallentire et al. 2019	3	4	2	3	3	2	3	2.9
Zira et al. 2020	3	3	2	3	2	3	3	2.7
Turner et al. 2023	3	4	2	3	2	3	2	2.7
Paris et al. 2022 using Scherer et al. 2018	3	4	3	2	2	1	3	2.6
Scherer et al. 2018*	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Leinonen et al. 2014	3	4	2	3	3	1	2	2.6
Boggia et al. 2019	3	4	2	3	1	2	2	2.4
Müller-Lindenlauf et al. 2010	3	3	1	2	2	3	2	2.3
Zucali et al. 2016	3	4	1	3	1	2	2	2.3
Weeks et al. 2016	2	4	1	3	2	1	2	2.1
Geß et al. 2020	3	3	1	3	1	1	2	2.0
Average Score	2.9	3.7	1.7	2.8	1.9	1.7	2.3	n/a

*Scherer et al. 2018 is scored through Paris 2022, as scoring the criteria twice for the same method would bias results

Figures

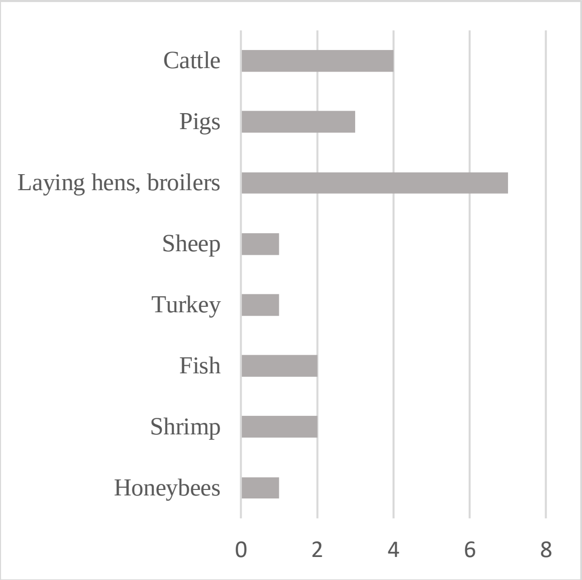


Figure 5 Number of papers covering different animal species

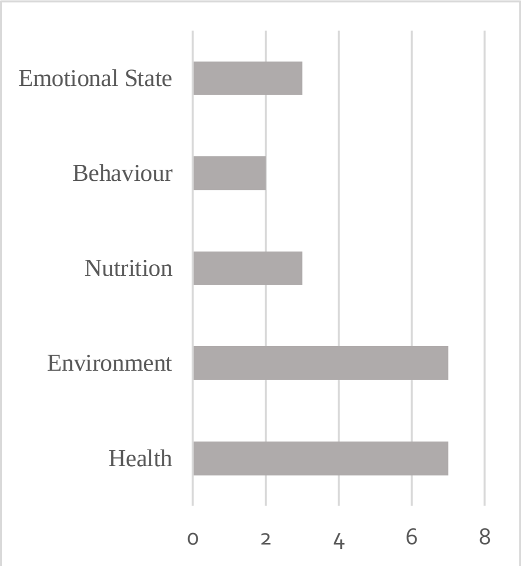


Figure 6 Number of papers covering different animal welfare domains

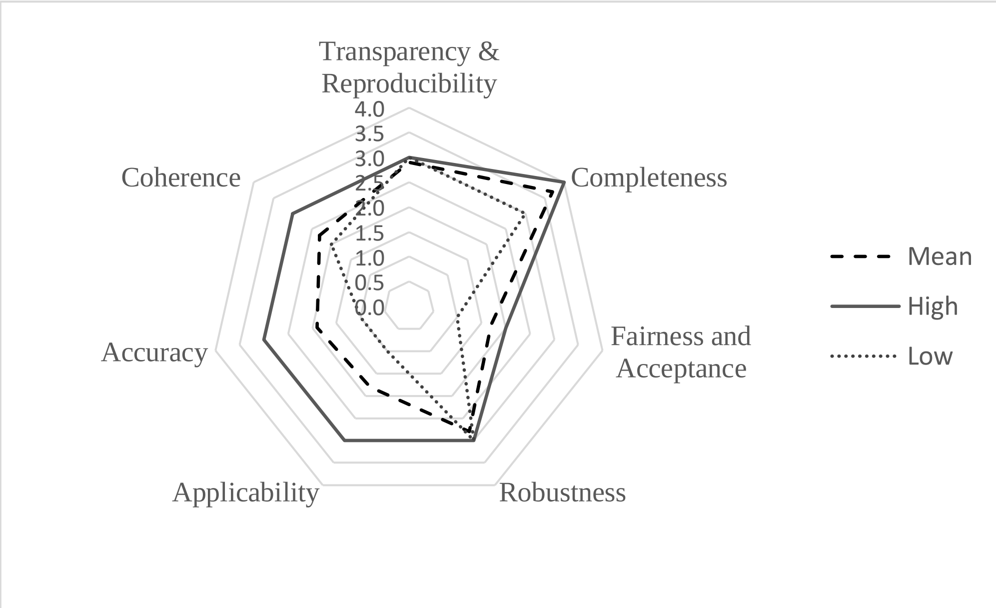


Figure 7 Scores of mean, highest and lowest performing papers across general and specific criteria

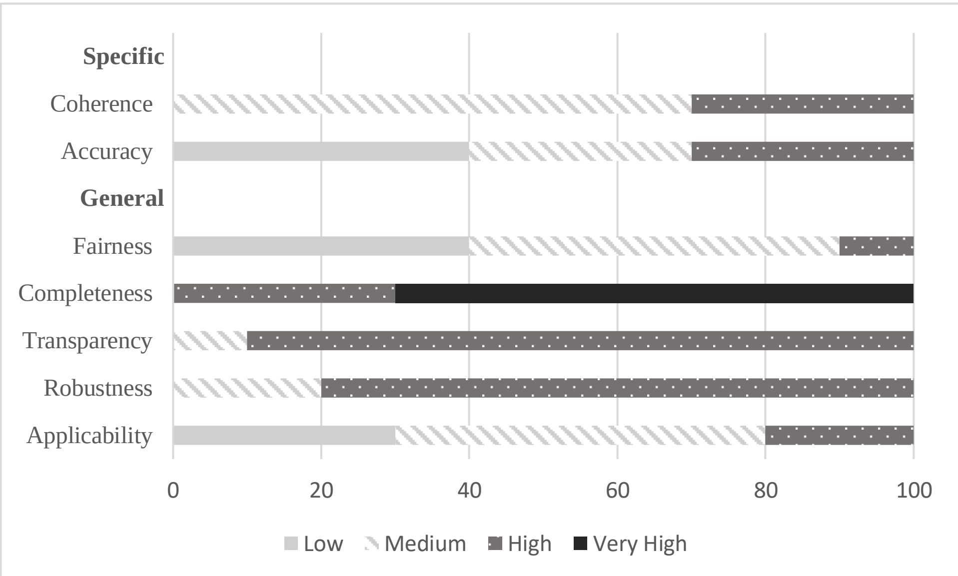


Figure 8 Distribution of scores for different criteria

SG4: Evaluating methods to include human nutritional aspects in Life Cycle Assessment for livestock systems and products

Klara van Mierlo[a], Daniel A. Mekonnen[a], Maxime Fossey[b], Simon Moakes[c,d], Bernando Valenti[e], Roline Broekema[a], Coen van Wagenberg[a], Gilles Nassy[f], Pietro Goglio[e]

[a]Wageningen Social and Economic Research, Droevendaalsesteeg 4, 6708 PB, Wageningen, The Netherlands
[b]Institut de l'élevage (IDELE), 149 rue de Bercy, 75012 Paris, France
[c]Research Institute of Organic Agriculture (FiBL), Frick, Switzerland
[d]IBERS, Aberystwyth University, UK
[e]Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy
[f]Institut du Porc (IFIP), La Motte au Vicomte, 35651 Le Rheu, France

Abstract

When assessing the environmental impact of livestock products, it is important to account for their function of supplying multiple important nutrients. Several methods to do this have been proposed in the literature, but little consensus exists on the choice of nutritional metric to use when integrating nutritional aspects in Life Cycle Assessment (LCA) of agricultural products. The goal of this study was therefore to identify the most appropriate method(s) to integrate nutritional parameters in LCA. By means of a systematic literature review and a screening step, relevant studies and related methods that integrate nutritional aspects in functional units (FU) were identified. The identified studies were scored based on the general criteria "transparency and reproducibility", "completeness", "fairness and acceptance" and "robustness", which were defined by means of a literature review of LCA frameworks and expert workshops. The identified methods were subsequently scored based on the specific criteria "coverage of multiple nutrients", "consideration of human nutritional requirements" and "accuracy", which were defined in expert workshops as well. Based on the latter, 16 high scoring methods were selected. Out of this list, 4 methods were deemed the most appropriate, based on their ability to include multiple nutrients in a FU and to calculate the nutrient scores objectively. Recommendations were formulated to test these methods in different contexts to identify the most appropriate method.

Introduction

Worldwide, demand for livestock products is predicted to double over the next decades due to population growth and increasing economic prosperity (Godfray et al., 2018). Further increases in production may put pressure on available resources such as land or water, and lead to higher GHG emissions and other environmental impacts (Van Zanten et al., 2018; Willett et al., 2019). Poor dietary patterns are responsible for many illnesses and deaths globally (Gakidou et al., 2017; Godfray et al., 2018), while healthy diets are unaffordable for nearly 3 billion people (Gaupp et al., 2021). However, livestock farming plays a vital role in food and nutrition security by providing several key components of the human diet, such as high quality protein (i.e. containing a favorable amino acid profile), vitamin B12, iron and zinc (Godfray et al., 2018; Mayer Labba et al., 2022; Mehrabi et al., 2020; Vieux et al., 2022; Wu et al., 2014).

The environmental impact as well as the nutritional profile of different livestock products differ considerably (Poore and Nemecek, 2018; Wu et al., 2014), For example, the climate change impact of beef is approximately ten times higher than poultry meat (Poore and Nemecek, 2018), while its iron content is approximately four times higher (RIVM, 2019). When comparing the environmental impact of different livestock products, such as meat-, dairy- and egg products, it is therefore important to also take into consideration the (primary) functions of these products, such as the supply of nutrients (Tounian, 2022), their ability for satiety and/or their human health effect (Weidema and Stylianou, 2020). In general, livestock products are a good source of protein, iron, vitamin B12, among others (Wu et al., 2014), but also can contain high content of saturated fat (Godfray et al., 2018). When comparing different livestock products, it is therefore important to consider a variety of nutrients, both qualifying (positive, desirable) and disqualifying (negative, undesirable) nutrients.

As livestock products contribute to environmental problems and at the same time deliver important nutrients (Godfray et al., 2018), a combined analysis of environmental impacts and nutritional factors is useful. Life Cycle Assessment (LCA) is a method to quantify environmental impacts of products and processes, considering their complete life cycle, from the exploitation of raw materials ("cradle") to the waste and/or recycling of end products ("grave"), considering multiple environmental problems (Hauschild et al., 2018). It has been applied to livestock products in multiple studies (Grossi et al., 2019; Poore and Nemecek, 2018).

The combined assessment of LCA and nutritional aspects can be approached in different ways, depending on the goal of the study. These approaches include analyses on both diet and product levels. Diet level studies include analyses that investigate the environmental consequences of changing from current diets to recommended diets (Behrens et al., 2017; Springmann et al., 2020), as well as studies that apply mathematical optimization models to find combinations of food products with lower environmental impacts and adequate nutritional supplies (Broekema et al., 2020; Tyszler et al., 2016).

Product level studies include the consideration of nutrition as an impact pathway to human health effects, as well as the inclusion of nutritional aspects in functional units in LCA's of food products (Jolliet, 2022; Weidema and Stylianou, 2020). The functional unit quantifies the function of the studied product(s) or process(es) for which the environmental impacts are quantified. It therewith serves as a reference for comparison, and should be clearly defined, particularly in comparative LCA's (ISO, 2006b). The supply of nutrients is an important function of food products, but it is not straightforward to grasp different nutrients in a single functional unit. However, several research studies tried to integrate nutritional aspects in the functional unit. Most research studies consider a single nutrient as functional units, for example the protein or omega-3 content of the product (Detzel et al., 2021, McNicol et al. 2024). When considering protein, not only the protein quantity is important, but also the quality, which is reflected in the amino acids composition of the proteins (McNicol et al. 2024). The fulfilment of human requirements for separate amino acids differs between protein sources and should therefore be looked at separately (FAO/WHO/UNU, 2007; Wu et al., 2014). Therefore, several studies have proposed to integrate protein quality in functional units, by multiplying the protein content with an amino acid score, which represent the protein quality of a certain product (Berardy et al., 2019; McAuliffe et al., 2022). Other research studies have integrated multiple nutrients in functional units, by considering nutrient density scores (Ridoutt, 2021a; Van Dooren et al., 2017). The Nutrient Rich Food (NRF) index, for example, is a metric for nutrient density, reflecting the contents of qualifying and disqualifying nutrients compared to their recommended amounts (Drewnowski, 2009).

Different levels of complexity in the definition of nutritional functional units are possible, as identified by McAuliffe et al. (2020): i) including a single nutrient, ii) including multiple nutrients in one functional unit and iii) including nutrients linked to dietary contexts, considering nutritional requirements. Related to each of these approaches, the following methodological issues were identified by the FAO: i) the selection of multiple relevant nutrients to reflect the function of the food products, ii) the quantification of nutritional values in relation to human nutritional requirements, which differ between men and women and age groups and activity levels, and considering the bioavailability of the nutrients, iii) the effect of processing and preparation steps on the nutritional value and environmental impact of food products, and iv) the variability in nutritional value and environmental impact of individual food products within certain food groups (e.g. "apple" in "fruit") (McLaren et al., 2021).

This study addresses these methodological issues by evaluating methods to include nutritional aspects in functional units in livestock LCA's on product level by means of a systematic literature review, following the harmonization approach presented by Goglio et al. (2023).

STUDY AIM

This study aimed at i) selecting the most appropriate methods to include nutritional parameters in livestock LCA's on product level, based on a systematic literature review following the harmonization approach presented by Goglio et al. (2023), ii) formulating recommendations for LCA practitioners, and iii) identifying further methodological developments for including nutritional aspects in LCA on product level.

Methods

GENERAL APPROACH

In our study we adopted the harmonization approach from the study of Goglio et al. (2023), which used the Delphi method, a participatory approach generally existing of multiple structured surveys with different stakeholders (Mullender et al., 2020). Literature reviews and workshops with experts (n=21) from different countries and research fields (LCA, biodiversity, nutrition and animal welfare) were executed.

Next to nutritional aspects, four other topics related to LCA of livestock products were defined by Goglio et al. (2023): i) food, feed, fuel and biomaterial competition, crop-livestock interaction, circular economy; ii) biodiversity; iii) animal welfare; iv) nutritional aspects; and v) greenhouse gas (GHG) emissions issues. For each key topic, a literature review was set up. Search and screening criteria for the literature searches were defined. Consequently, general (similar for each key topic) and specific criteria (different for each key topic) for the method evaluations were defined. In this report, we only present the results for the nutritional aspects, the other aspects are reported in different reports.

In this study, a literature review was applied to the key topic of nutritional aspects. The following steps were followed: i) identification of studies by means of a structured literature search, ii) selection of relevant studies based on a screening step, iii) scoring of selected studies based on general criteria, iv) identification of methods in selected studies, v) evaluation of methods based on specific criteria, vi) selection of high scoring methods based on the specific criteria, vii) identification of the most appropriate methods, based on previous steps.

LITERATURE SEARCH, SCREENING AND RELEVANT PAPER SELECTION

A variety of approaches have been proposed to integrate nutrition into LCA (McLaren et al., 2021). In this study the focus was on including nutritional aspects in the definition of the functional unit of livestock LCA's. The search criteria therefore combined terms related to LCA with terms related to nutrients and livestock products (Table 1). These search terms were used to search three commonly used and complementing databases in the field of environmental sciences (Adriaanse and Rensleigh, 2011): Web of Science (WoS), Scopus and Google Scholar. Because of differences in algorithms behind the databases, different search terms were used in each database, to obtain approximately the same number of studies. In Google Scholar, broad terms were entered, and the first 30 relevant studies were selected.

To map recent developments, studies published from 2012 to March 2022 were selected. Roux (2021) carried out a literature review to functional unit selection in agrifood sector (unpublished work). As this review lists relevant references, they were added to the final list of studies. Finally, the study of McLaren et al. (2021) was added to the list, together with a few studies already available to the involved reviewers (McAuliffe et al., 2018; Nassy et al., 2021; Ridoutt, 2021b).

Identification of relevant studies and methods included the following three steps: 1) removing duplicates in the identified studies in the different databases; 2) screening the title and abstract for relevance. In this step, the following studies were excluded: review studies, studies in which no actual LCA was executed, diet level studies, studies that did not consider (human) nutritional aspects; 3) analysing the whole text of each relevant paper to identify the methods used to estimate the (human) nutritional aspects.

Table 1. Combinations of search terms for the topic "nutritional aspects"

No.	Database	Date of search	Search terms part 1	Search terms part 2	Search terms part 3	Search terms part 4
1	WoS / Scopus	21/03/22	("Life cycle assessment" OR " life cycle analysis") AND	("nutrition*" OR "protein" OR "amino acid*" OR "protein quality" OR "B12" OR "iron" OR "calcium" OR "vitamin D" OR "calcium") AND	("protein source*" OR "livestock" OR "dairy" OR "cattle" OR "sheep" OR "pig*" OR "poultry" OR "goat*" OR "farm*" OR "agricultur*" OR "milk" OR "cheese" OR "butter" OR "meat" OR "ham" OR "bacon" OR "pork" OR "beef" OR "lamb" OR "egg*" OR "leather" OR "chicken" OR "cow" OR "husbandry" OR "rearing") AND	"functional unit"
2	WoS/ Scopus	8/03/22	("Life cycle assessment" OR " life cycle analysis" OR "nutritional life cycle assessment") AND	("nutrition*" OR "nutritional functional unit") AND	("protein source*" OR "livestock" OR "meat*" OR "beef" OR "cow" OR "veal" OR "pork" OR "porc" OR "pig" OR "lamb" OR "mutton" OR "horse" OR "goat" OR "poultry" OR "chicken" OR "turkey" OR "duck" OR "goose" OR "game" OR "pancetta" OR "sausage" OR "*burger" OR "mortadella" OR "salami" OR "pate" OR "ham" OR "bacon" OR "foie gras" OR "schnitzel" OR "dairy" OR "milk" OR "butter" OR "yohgurt" OR "yogurt" OR "*cheese" OR "cream" OR "pudding" OR "egg*")	
3	Google scholar	28/02/22	"LCA" AND	"nutritional" AND	"livestock production"	
4	Google scholar	10/03/22	"LCA" AND	"nutritional functional unit"AND	("beef" OR "Pork" OR "Chicken" OR "Lamb" OR "Meat" OR "egg" OR "dairy" OR "Fat" OR "Energy" OR " Protein")	

5	References of literature review of Roux (2021)	28/02/22
6	Additional studies available to reviewers (McAuliffe et al., 2018; McLaren et al., 2021; Nassy et al., 2021; Ridoutt, 2021b).	

CRITERIA TO EVALUATE IDENTIFIED LCA STUDIES AND METHODS

The various LCA studies discovered in the literature review (Section 2.2) were initially evaluated using general criteria defined by Goglio et al. (2023), such as transparency and completeness, which are applicable to any LCA study.

Following the evaluation using general criteria, three specific criteria were selected to evaluate methods that include nutritional aspects in LCA: coverage of multiple nutrients, consideration of human nutritional requirements, and accuracy.

Coverage of multiple nutrients

On the one hand, the function of food products in human nutrition relates to the supply of multiple qualifying (positive, desirable) nutrients. For livestock products, these include protein, zinc, calcium, iron and vitamin B12 (Van Zanten et al., 2018), among others. On the other hand, livestock products also contain disqualifying (negative, non-desirable) nutrients, such as sodium and saturated fat, as reflected in nutrient density scores (Drewnowski, 2009). Therefore, the specific criteria 'Coverage of multiple nutrients' was proposed to capture the wide nutritional functions of livestock products in functional units (Table 2). The following scoring scale was proposed, with a higher score indicating that the method performed better on this criterion (Table 2):

- score 1: the LCA method uses a weight based functional unit, no nutritional parameters considered,
- score 2: the LCA method considers a single nutrient or multiple nutrients in separate functional units,
- score 3: the LCA method considers nutrient density scores including multiple – both qualifying and disqualifying - nutrients in one functional unit.

Consideration of human nutritional requirements

The specific criterion "Consideration of human nutritional requirements" aims at assessing if human nutritional requirements are captured in the LCA method with regards to the functional unit, as the function of food products are not only related to the content of nutrients in food products, but also to human nutritional requirements (McAuliffe et al., 2016) (Table 2). Specifically, both the minimum and maximum human requirements for nutrients should be considered, where such limits are defined. The following scoring scale was proposed, with higher scores indicating better performance. (Table 2):

- score 1: human nutritional requirements are not considered in the LCA,
- score 2: minimum or maximum human nutritional requirements are considered in the LCA,

- score 3: both minimum and maximum human nutritional requirements considered in LCA.

Accuracy

The specific criterion "Accuracy" in the context of nutritional aspects is calculated as the average of coverage and consideration and can thus be seen as a summary criterion (Table 2).

STUDY AND METHOD SCORING AND SELECTION OF MOST APPROPRIATE METHODS

Each selected study was scored based on the general criteria and each method on the specific criteria. Initial scoring of the studies based on general criteria was performed by five LCA and nutritional experts, and this scoring was subsequently checked by one expert. Each method was scored based on the specific criteria independently by two reviewers. After both reviewers had finished scoring all methods, any deviations in scores between the reviewers was discussed by them and a consensus score was achieved, through targeted discussions (Macombe et al., 2018; Mullender et al., 2020). The methods with the highest scores (2.5 or higher) for the specific criterion accuracy were selected and considered for further evaluation. In a last step, the most appropriate methods to include nutritional aspects in LCA were selected, based on their ability to include multiple nutrients in one functional unit, their inclusion of nutrient scores that can be calculated objectively and their compliance with most of the recommendations by the European Food Safety Authority regarding eligibility to health and nutrition claims (EFSA, 2008).

Table 2. Specific criteria to evaluate studies and methods

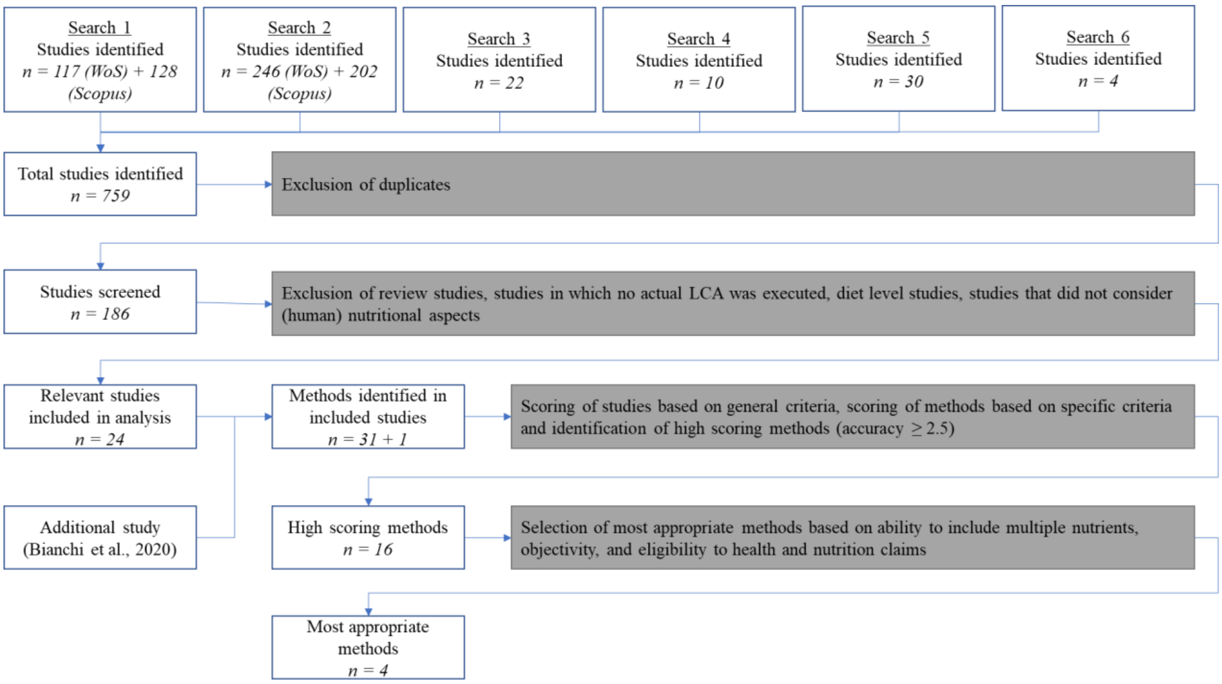
Specific criteria definition	Score 1	Score 2	Score 3
Coverage of multiple nutrients: aims to capture the wide nutritional functions of livestock products in functional units.	The LCA employs weight based functional unit, no nutritional parameters considered	The LCA method considers a single nutrient or multiple nutrients in separate functional units	The LCA method considers nutrient density scores including multiple nutrients in one functional unit
Consideration of human nutritional requirements: aims at assessing if human nutritional requirements are captured in the LCA method with regards to the functional unit, as the function of food products are not only related to the content of nutrients in food products, but also to human nutritional requirements (McAuliffe et al., 2016). Specifically, both the min and max human requirements for nutrients should be considered.	Human nutritional requirements are not considered in the LCA	Human nutritional requirements are considered in the LCA	Both minimum and maximum human nutritional requirements are considered in the LCA
Accuracy: aims at the consideration of differences in nutritional requirements and in nutritional compositions which are captured by the coverage of multiple nutrients and consideration of nutritional requirements criteria.	Calculated as the average of coverage and consideration and can thus be seen as a summary criterion.		

Results

NUMBER OF IDENTIFIED, SCREENED, SCORED AND SELECTED STUDIES

As shown in Figure 1, the literature search presented in Table 1 resulted in a total of 759 studies. After removing duplicates, 186 studies were left. After screening the studies, 24 relevant studies were found, containing a total of 31 methods. During the review, one more study (Bianchi et al., 2020) was added to the list before the methods were evaluated using specific criteria, resulting in a final list of 25 studies and 32 methods. The studies were scored based on the general criteria, and the methods were scored based on the specific criteria. Based on the specific criteria "accuracy", 16 high scoring methods (≥2.5) were retained. In a last step, the 4 most appropriate methods to include nutritional aspects in LCA were selected, based on based on their ability to include multiple nutrients in one functional unit, their inclusion of nutrient scores that can be calculated objectively and their compliance with the recommendations by the European Food Safety Authority regarding eligibility to health and nutrition claims (EFSA, 2008).

Figure 1. Results of the selection of studies and methods with nutritional aspects in LCA.



SCORING OF THE SELECTED STUDIES AND METHODS

Each study was evaluated and scored in terms of the general criteria and on the specific criteria. Table 3 shows the selected studies and methods and their scores based on the general and specific criteria, respectively. The distribution of the scores for general criteria applied to the studies are presented in Figure 2. More than half of the methods scored 2 and above (out of 4) for each of the five components. In a next step, the methods were evaluated and scored in terms of the specific criteria, i.e. coverage, consideration, and accuracy (Figure 3). In terms of accuracy of the method, i.e. the average of the scores on coverage and consideration, 22.6% of methods had a score of 3 (out of 3), 32.3% a score of 2.5, and about 45.2% a score of 2 or lower.

Table 3. Scoring of selected studies and methods with nutritional aspects in LCA.

		General criteria: paper assessment					**Specific criteria: method assessment**		
	Method name	Transparency and Reproducibility	Completeness	Fairness and Acceptance	Robustness	Applicability	Coverage of multiple nutrients	Consideration of human requirements	Accuracy
Bruno et al. (2019)	Calorie intake	3	3	2	2	3	2	1	1.5
Salazar et al. (2019)	n3FA content	2	4	3	3	1	2	2	2
Jan et al. (2019)	Digestible energy	1	4	2	2	1	2	1	1.5
Berardy et al. (2019)	Protein quality	1	2	3	3	3	2	1	1.5
Schaubroeck et al. (2018)	Nutritional score	3	4	3	3	1	3	3	3
Van Mierlo et al. (2017)	Linear programming	3	4	3	3	1	3	1	2
Schmitt et al. (2016)	Nutrient content	1	2	3	3	3	2	1	1.5
Doran-Browne et al. (2015)	NRF9.3	2	2	2	2	2	3	3	3
Tello et al. (2021)	Impact 2002+	2	3	1	3	1	3	1	2
Mogensen et al. (2020)	Protein, energy content	2	2	1	3	2	2	1	1.5
Röös et al. (2020)	Nutrient content	1	1	2	2	1	2	3	2.5
Walker et al. (2019)	Recipe 2016 (DALYs)	2	3	2	3	1	1	1	1
Hitaj et al. (2019)	Diet model	2	2	3	2	2	1	2	1.5
Fresán et al. (2019)	Source of protein	3	3	1	3	2	2	1	1.5
Van Dooren et al. (2017)	SNRF3.3	2	1	2	2	2	3	3	3
Masset et al. (2015)	SAIN,LIM	2	2	3	2	2	3	3	3
	UK Ofcom	2	2	3	2	2	3	3	3
Masset et al. (2014)	SAIN,LIM	3	2	2	3	2	3	3	3
Oonincx and De Boer (2012)	Edible protein	3	2	1	2	3	2	1	1.5
Xu et al. (2017)	Nu21	1	1	2	2	2	3	2	2.5
	Nu5						3	2	2.5
Xu et al. (2020)	Nu11	3	3	2	2	2	3	2	2.5
Green et al. (2021)	NRF21.2	3	3	2	3	2	3	2	2.5
	NRFprotein sub-score						3	2	2.5
Smith et al. (2022)	DALY model	1	2	3	2	1	2	2	2
McAuliffe et al. (2018)	UKNIprot7	2	2	3	3	2	3	2	2.5

	UKNIprot10						3	2	2.5
	UKNIprot7-2						3	2	2.5
	UKNIprot10-2						3	2	2.5
Ridoutt (2021b)	NRF-ai, others	3	4	2	2	2	3	1	2
Bianchi et al. (2020)*	NRF11.3						3	3	3

* Not scored on the general criteria

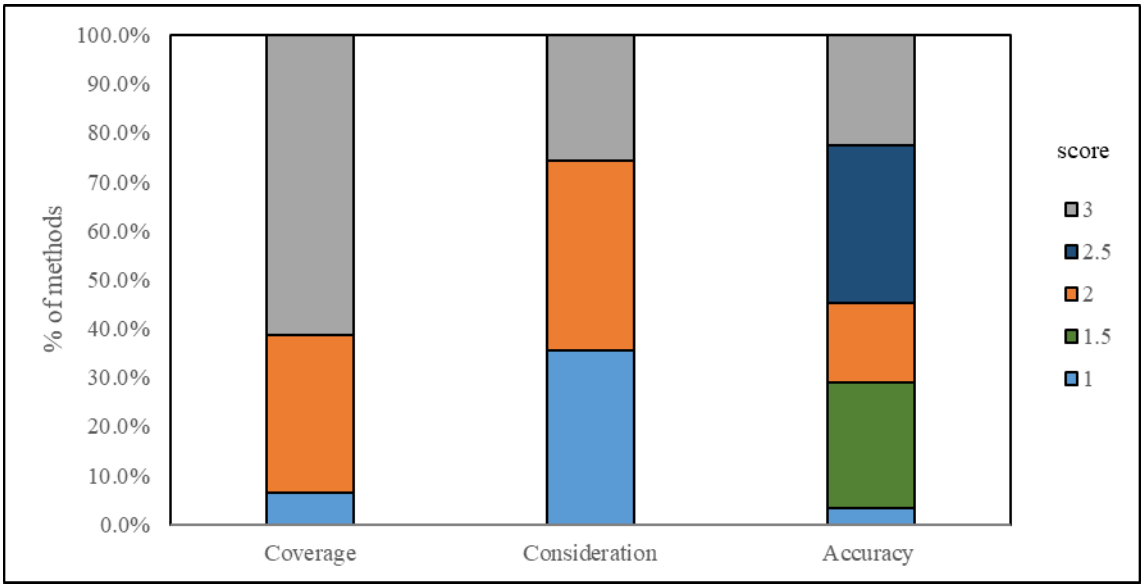


Figure 2.Distribution of scores on the general criteria of the 24 selected studies (% shares)

.

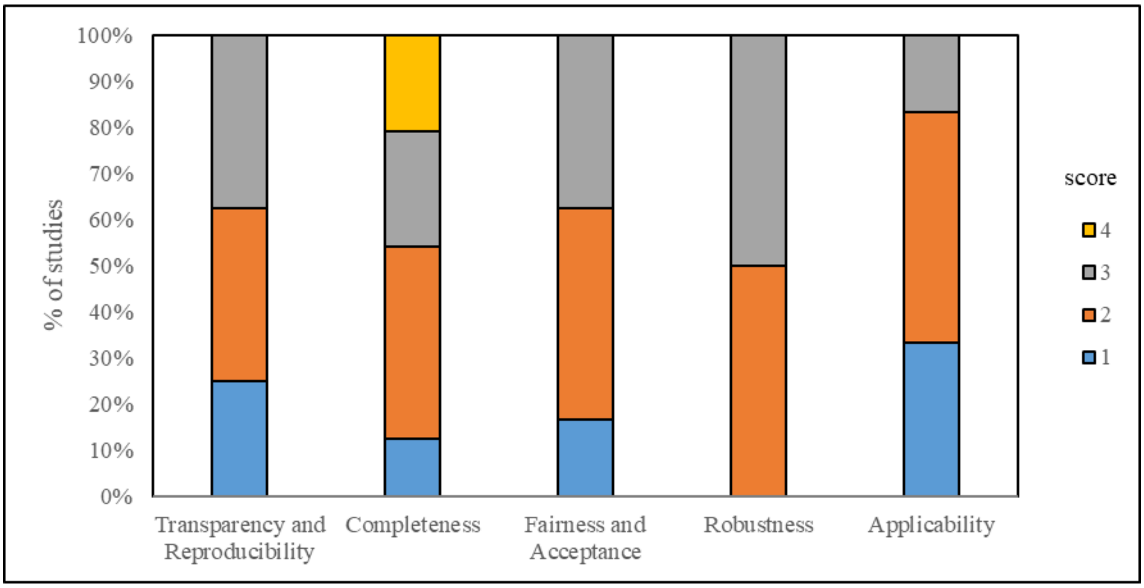


Figure 3. Distribution of scores on the specific criteria of the 32 selected methods (% shares)

SELECTION OF HIGH SCORING METHODS

The methods that scored 2.5 and higher for the specific criterium accuracy were selected. Table 4 describes the selected methods, including their FUs, the total number of nutrients included, and the types of nutrients – i.e. qualifying and disqualifying nutrients.

Among the seven methods that received an average score of 3, three were variants of the Nutrient Rich Food (NRF) index, two were based on SAIN,LIM, one based on the UK Ofcom, and one method based on the nutritional score of a meal on meeting nutritional criteria. Each of these are further discussed in more detail below.

Nutrient Rich Food (NRFx.y) index: The NRF index is calculated from x number of qualifying nutrients and y number of disqualifying nutrients (See Table 4). Calculation of the index takes into account the dietary reference intake (DRI) of qualifying nutrient and the maximum recommended intake of disqualifying nutrient (MRI) (Bianchi et al., 2020). The nutrient density is measured as the sum of percentage daily values of x qualifying nutrients, minus the sum of percentage maximum recommended values for y disqualifying nutrients, calculated per reference amount and capped at 100% of the recommended intake. The NRF method is flexible in that the number of qualifying nutrients can vary depending on context (see Table 4 for specific nutrients in $NRF_{9.3}$, $NRF_{3.3}$, $NRF_{11.3}$, $NRF_{21.2}$, $NRF_{11.3}$), thus adapting to the level of coherence needed in line with dietary guidelines. Furthermore, with the purpose of differentiating the contribution of individual nutrients to the final score, capping and weighting are applied to nutrient indexes. According to Bianchi et al., (2020), capping is used to avoid over-crediting nutrient contents that exceed their DRI (applicable only to qualifying nutrients except for fibre and omega-3 fatty acids whose DRIs are indicated as minimum recommended amounts to be introduced in the diet) by rounding off their nutrient content per reference unit to 100% of DRI. Similarly, weighting is used to give different weights to nutrients based on the intake status in the given population (Bianchi et al., 2020). That is, qualifying nutrients receive a higher weight when nutrient intake in the average population is sub-optimal, and lower weight when intake exceeds DRIs. Whereas disqualifying nutrients receive a higher weight when the population average intake exceeds the MRI, but they are capped to 100% of MRI when intake is lower than this value. The nutrient density can be calculated for three reference units: 100 g, 100 kcal, or the portion size, and calculations are based on the following equation (Bianchi et al., 2020):

$$NRF_{x,y} = \sum_{i=1}^{x} \frac{nutrient_i}{DRI_i} - \sum_{j=1}^{y} \frac{nutrient_j}{MRI_j}, \qquad (1)$$

where x is the number of qualifying nutrients, y is the number of disqualifying nutrients, $nutrient_i$ and $nutrient_j$ are the content of nutrient *I* or *j* per reference unit of the food product, DRI_i is the dietary reference intake of qualifying nutrient *i*, and MRI_j is the maximum recommended intake of disqualifying nutrient *j* (Bianchi et al., 2020). A higher value of $NRF_{x,y}$ means a higher nutritional quality. This method is widely used and has potential for further consideration.

.

Table 4. Methods that include nutritional aspects in LCA that had at least a score of 2.5 on accuracy.

Reference	Method	Functional unit	Cov e-rage	Consid er-ation	Acc ur-acy	# nu-trients	Qualifying (positive/desirable) nutrients	Disqualifying (negative/non-desirable) nutrients
Masset et al. (2015)	SAIN,LIM	SAIN per 100kcal and the LIM per 100g	3	3	3	8	Protein, fibre, calcium, iron, vitamin C (Vitamin D is optionally used when its content/recommendation ratio is greater than one of the basic nutrients)	Saturated fatty acids, added sugars, sodium
	UK Ofcom	Ofcom score per 100 g of food	3	3	3	9	Protein, fibre (and fruit, vegetable, nut content)	Saturated fat, sodium, total sugar, energy
Doran-Browne et al. (2015)	NRF$_{9.3}$	Nutrient density per 100 g	3	3	3	12	Protein, fibre, calcium, iron, vitamin A, C, E, magnesium, potassium	Saturated fat, sodium, added sugar
Van Dooren et al. (2017)	SNRF$_{3.3}$	Nutrient density per 100 g	3	3	3	6*	Essential fatty acids (EFA), plant protein, dietary fiber	Saturated fatty acids, sodium, added sugar
Schaubroeck et al. (2018)	Nutritional score of meal based on nut. criteria	Nutrient content in g per 100 kcal	3	3	3	7	Protein, fat, carbohydrate, energy,	Saturated fat, salt, sugar
Bianchi et al. (2020)	NRF$_{11.3}$	Nutrient density per 100 g	3	3	3	14	Protein, fiber, calcium, iron, vitamin A, C, D, E, folate, magnesium, potassium	Saturated fat, added sugar, sodium
Xu et al. (2017)	Nu21	Nutrient density per kg of food	3	2	2.5	21	Protein, fat, carbohydrate, dietary fiber, calcium, iron, retinol, vitamin B1, B2, B3, C, E, potassium, sodium, magnesium, manganese, zinc, copper, phosphorus, selenium, and cholesterol	
	Nu5	Nutrient density per kg of food	3	2	2.5	5	Protein, dietary fiber, carbohydrate, vitamin B1, B2	
Xu et al. (2020)	Nu11	Nutrient density per kg of food	3	2	2.5	11	Protein, dietary fiber, carbohydrate, calcium, iron, vitamin B1, B2, B3, magnesium, zinc, potassium	
Green et al. (2021)	NRF$_{21.2}$	Nutrient density considering energy content	3	2[$]	2.5	23	Protein, polyunsaturated fat, carbohydrate, fiber, calcium, iron, vitamin A, B1, B2, B3, B6, B12, C, potassium, phosphorus, copper, zinc, folate, choline, manganese, magnesium	Sodium, saturated fat

	NRFprotein sub-score	Nutrient density considering energy content	3	2$	2.5	5	Calcium, iron, vitamin B12, riboflavin,	Saturated fat
Röös et al. (2020)	Averge nutrient intake	Separate functional units for each nutrient	2	3	2.5	10	Protein, fat, carbohydrate, fiber, iron, vitamin B12, zinc, folate, selenium, energy	
McAuliffe et al. (2018)	UKNIprot7	Nutrient content per 100g of meat	3	2	2.5	7	Protein, monounsaturated fatty acids, EPA+DHA, calcium, iron, vitamin B2, folate	
	UKNIprot10	Nutrient content per 100g of meat	3	2	2.5	10	Protein, monounsaturated fatty acids, EPA+DHA, calcium, iron, vitamin B2, B12, folate, selenium, zinc	
	UKNIprot7-2	Nutrient content per 100g of meat	3	2	2.5	9	Protein, monounsaturated fatty acids, EPA+DHA, calcium, iron, vitamin B2, folate	Sodium, saturated fatty acids
	UKNIprot10-2	Nutrient content per100g of meat	3	2	2.5	12	Protein, monounsaturated fatty acids, EPA+DHA, calcium, iron, vitamin B2, B12, folate, selenium, zinc	Sodium, saturated fatty acids

* Based on only macro nutrients but follows same approach as $NRF_{9.3}$; $ Note: the minimum values for each of disqualifying nutrients are set to 0, but unlike other NRF varieties (such as $NRF_{9.3}$) the maximum value was capped at 100% for the aggregate result instead of each of individual nutrients. Hence, consideration was scored as level 2. Nu refers to the sum of the relative values of the studied nutrient elements in the food, calculated as the sum of the daily reference intake (RDI) value of select nutrients; UKNIprot is a nutrient index that simply rewards foodstuffs with higher contents of select qualifying nutrients as %RDI and penalize disqualifying nutrients where applicable.

SAIN,LIM5.3: The SAIN,LIM index is based on two components of nutritional scores, the SAIN (score for the nutritional adequacy of qualifying nutrients) and the LIM (score for disqualifying nutrients). Each score is calculated as an average (nutrient content/recommendation) ratio. Nutritional quality of foods is integrated into one dimension by taking the ratio of SAIN/LIM (where LIM is set to one when lower than one), and higher value means higher nutritional quality (Masset et al., 2015). The SAIN,LIM$_{5.3}$ scores are calculated in two parts, with the SAIN per 100 kcal (or 420 kJ) and the LIM per 100 g (Darmon et al., 2009). The SAIN score is an unweighted arithmetic mean of the percentage adequacy for 5 positive nutrients (plus one optional nutrient):

$$SAIN_i = \frac{\sum_{p=1}^{p=5} ratio_{i,p}}{5} \times 100 \,, \qquad (2)$$

with $ratio_{i,p} = [\frac{nutrient_{i,p}}{RV_p}] \times \frac{100}{E_i}$, (3)

where $nutrient_{i,p}$ is the quantity (g, mg, or lg) of positive nutrient p in 100 g of food i, RV_p is the daily recommended value for nutrient p, and E_i is the energy content of 100 g of food i (in kcal/100 g). Depending on lipid contents of individual foods (for example for foods providing >97% of their energy as lipids), optional nutrients including vitamin E, linolenic acid, and monounsaturated fatty acids can be used to replace up to two of the five basic nutrients including protein, fibre, calcium, vitamin C, and iron. Similarly, Vitamin D can be used as optional nutrient for foods providing <97% of their energy as lipids)

The LIM score is the mean percentage of the maximal recommended values for 3 negative nutrients (saturated fatty acids, added sugars, and sodium):

$$LIM_i = \frac{\sum_{l=1}^{l=3} ratio_{i,l}}{3} \,, \qquad (4)$$

with $ratio_{i,l} = \left[\frac{nutrient_{i,l}}{MRV_l}\right] \times 100$ (5)

where $nutrient_{i,l}$ is the content (g, mg) of limited nutrient l in 100 g of food i, and MRV_l is the daily maximal recommended value for nutrient l. The method has potential for further consideration.

UK Ofcom: The UK Ofcom nutrient profiling index scores food and drinks separately (on nutrient scale per 100 g of food) to define products that are 'healthy' (or at least not 'unhealthy') and foods that are 'less healthy' (Rayner et al., 2009). The UK Ofcom incorporates both positive and negative nutrients on a single scale. Both qualifying and disqualifying nutrients are scored (by means of 'A' points and a 'C' points respectively), in which higher scores indicate an unhealthier food/drink (Rayner et al., 2009). The overall score is calculated with the following formula:

Overall score = total A points minus total C points, in which

- Total 'A' points = [points for energy] + [points for saturated fat] + [points for sugars] + [points for sodium]
- Total 'C' points = [points for fruit, vegetables and nut content] + [points for fibre (either NSP or AOAC)] + [points for protein]

The scale for A points goes from 0 to 10, with a higher score indicating a higher content of the nutrients. The scale for C points goes from 0 to 5, with a higher score indicating a higher content of the ingredients/nutrient. Food products with a score of 4 points or more and drinks with a score of 1 point or more are considered as less healthy (Rayner et al., 2009). As the scoring is simply based on two opposite ordinal scale, not quantitative, which poorly combines together in a single score, this method was less preferred for further consideration.

Nutritional Scoring of a Meal: This method scores meals based on seven nutritional parameters not being in line with nutritional guidelines (see Table 4 for type of nutrients considered). The method assigns one penalty point for each parameter which is not in line with a predefined criterion (e.g. protein content is lower than x g per day). The nutritional score is then calculated by summing over the nutritional score of a meal for seven macro nutrients plus one, according to the formula: $Nutritional\ score\ of\ meal\ =\ 1\ +\ number\ of\ nutritional\ criteria\ not\ met$; yielding a score from 1 to x + 1, with x being the number of criteria for seven nutrients considered (Table 4). The resulting score indicates how many criteria are not met, and a higher value means lower nutritional quality. For instance, a meal with a score of 8 failed on all criteria, while a meal with a score of 1 meets all criteria for the investigated nutritional parameters (Schaubroeck et al., 2018). Even though this method received one of the highest scores based on the specific criteria (i.e. coverage and consideration), this method is not preferred for further consideration. This is because we aim for methods measuring nutritional aspects in LCA on product level rather than on meal level, even though the authors noted that the method could also be applied to products level. Further, there is doubt whether nutrient content of a meal can be accurately measured using this method.

SELECTED MOST APPROPRIATE METHODS

Various methods in LCA studies that integrate nutritional aspects in functional units were evaluated based on pre-defined specific criteria, including coverage of multiple nutrients, consideration of human nutritional requirements, and accuracy as described above. Among methods that were scored the highest, the SAIN,LIM and NRF are the most appropriate methods for including nutritional aspects in LCA on product level. In both of these methods, multiple nutrients can be considered in one functional unit, nutrient scores of foods can be calculated objectively, and technically calculation of the nutrient indices appears to be straightforward (Drewnowski, 2009; Fulgoni et al., 2009). In addition, the SAIN,LIM system complies with most of the recommendations by the European Food Safety Authority regarding eligibility to health and nutrition claims (Darmon et al., 2009; EFSA, 2008). The same can be argued about the NRF method varieties, since all of the nutrients considered in SAIN,$LIM_{5.3}$ are also part of the $NRF_{5.3}$, $NRF_{9.3}$ or $NRF_{11.3}$, even though no clear justifications are provided regarding selection of nutrients in the NRF methods. In addition, results of studies testing variants of the SAIN,LIM with other numbers of nutrients suggest that the ability of a given food to facilitate or impair the fulfilment of a large number of nutrient recommendations in a diet can be predicted on the bases of few nutrients only (Darmon et al., 2009). For example, the SAIN score comprising of only five nutrients was found to be highly correlated with SAIN scores comprising of a large number of important nutrients in foods, even though earlier studies using SAIN,LIM considered up to 23 different nutrients (Darmon et al., 2009). Similarly, Masset et al. (2015) reported that the SAIM,LIM ratio based on 5 positive and 3 negative nutrients correlates well with modelled diets that meet a full set of nutritional recommendations. Similarly, among 45 variants of the nutrient density index NRF (including $NRF_{9.3}$, $NRF_{11.3}$, and $NRF_{21.3}$), Bianchi et al. (2020) found that a Sweden-tailored $NRF_{11.3}$ index, calculated per portion size or 100 kcal with the application of weighting, ranked foods most coherently with the Swedish nutritional guidelines. So, assessing the nutritional aspects of diets in LCA's with the NRF method could potentially be conducted without the need for information on many nutrients. However, the impact of nutrient exclusion from NRF on the relative ranking of food products without diet consideration needs to be tested, for example by comparing the ranking of food products based on NRF with few nutrients to the ranking based on NRF with more nutrients. Bianchi et al. (2020) stress that selection of nutrients to be considered should be optimized for their level of coherence with dietary guidelines.

Considering the above, the following variants of the SAIN,LIM and NRF, including different (less and more) numbers of qualifying and disqualifying nutrients, were selected as the most appropriate methods to include nutritional aspects in LCA on product level:

1) $SAIN,LIM_{5.3}$ (Score for the nutritional adequacy of qualifying nutrients, Score for disqualifying nutrients)
2) $NRF_{9.3}$ (Nutrient Rich Foods)
3) $NRF_{11.3}$ (Nutrient Rich Foods)
4) $NRF_{5.3}$ (using same nutrients as in $SAIN,LIM_{5.3}$)

These methods have not been assessed using the same context. While the SAIN,LIM method complies with most of the recommendations by the European Food Safety Authority (EFSA, 2008), the method was tested based only on a typical French diet (Darmon et al., 2009). Similarly, much of the NRF varieties omit important nutrients such as vitamin B12 and selenium, and as a result tend to favour plant-based products to comparable products of animal origin (McAuliffe et al., 2020). Therefore, we cannot determine which of these is most preferred. Hence, we recommend using the four methods next to each other in the same context of livestock products and assess method sensitivity of the results (ISO, 2006a; ISO, 2006b). Comparing the results of the different selected methods might reveal limitations and/or facilitate selecting the most appropriate method(s), or a combination thereof.

Discussion and future research needs

Drawing on previous studies, the scientific group of the UN Food Systems Summit 2021 defined a nutritious food as "one that provides beneficial nutrients (e.g. protein, vitamins, minerals, essential amino acids, essential fatty acids, dietary fibre), and minimizes potentially harmful elements (e.g. anti-nutrients, quantities of sodium, saturated fats, sugars)." However, nutritious foods need to be produced, acquired, and consumed in a sustainable manner. According to Saarinen et al. (2017), for example, eating smaller portions of higher quality products may improve human nutrition whilst simultaneously reducing the global carbon footprint through potentially decreased food production. Hence, one of the steps to improve sustainability is through understanding the environmental impact of different foods that provide essential nutrients. Such understanding can help to design strategies that promote sustainable behaviour among consumers and producers. In this regard, integration of the nutritional quality of foods in LCA's has been found to be a promising approach (Bianchi et al., 2020; McAuliffe et al., 2022). However, there is little consensus on the choice of nutritional metric to use when integrating nutritional aspects in LCA (McAuliffe et al., 2020).

Our study found that the SAIN,LIM and NRF appear to be most appropriate metrics to include nutritional parameters in the functional unit of livestock LCAs. We recommend testing these methods in the same context to validate their applicability and to select the appropriate nutrients. While our analysis focused on integrating nutritional aspects in LCAs of livestock products, the selected methods are also applicable to other food products. This could be particularly useful when comparing livestock products with plant-based protein sources, which differ both in environmental and nutritional profile (Poore and Nemecek, 2018; Springmann et al., 2016).

In addition to the above, we propose several recommendations concerning the integration of human nutrition within agri-food life cycle assessments:

1. when combining nutritional and environmental factors , we recommend considering impacts at a consumption level. In this context, it is important to consider the effect of food processing and cooking processes on the nutritional values of the products (D'Evoli et al., 2009; Gómez et al., 2020).
2. We recommend reporting the nutritional profile of the studied products in combination with the main results of environmental impact per nutritional functional unit. This is important because the details on the full nutritional profile of the studied product is lost when expressing the environmental impact in terms of a single nutritional score (Ridoutt, 2021b).
3. We recommend considering nutritional guidelines and current food consumption patterns for selecting nutrients to be included in the nutritional scores. Even though nutrient density of products provides important information in combination with environmental performances of food products, a higher number of nutrients in the nutrient density calculation is not necessarily desirable. More robust solutions for nutrient selection are to compare multiple

food products in the context of an over/under supply of particular nutrients in current diets (McAuliffe et al., 2020), as foods rich in one nutrient may be lacking in another, and vice-versa, and there is no single function which is relevant to all foods (Ridoutt, 2021b). The decision to include or exclude nutrients in nutritional metrics is not always justified yet has the potential to greatly impact study results and conclusions. A second weakness is the relative ubiquity of individual nutrients in the food system. Some nutrients are widely available and sufficient intake is usually achieved regardless of specific food choices. These nutrients are less important from a public health nutrition perspective compared to those nutrients that are more widely under consumed (Ridoutt, 2021a). To resolve these weaknesses, Ridoutt (2021a) developed an alternative NRF index, the NRF-ai, which takes into account current shortages and excessive intakes of nutrients in the Australian population. This index includes all nutrients for which Estimated Average Requirements (EAR's) are defined by the National Health and Medical Research Council in Australia, and weighting factors are appointed to the separate nutrients based on their excess or shortages in current intake (Ridoutt, 2021a). In further research, when applying the SAIN,LIM and NRF in the same context, the approach of Ridoutt (2021a) could be followed to select and/or give weight to nutrients, based to on the background diet (shortages/excesses of nutrients) of the country/region where the product under study in consumed. Furthermore, nutritional indexes do not directly reflect the health effects of the studied foods. Saturated fat for example, which is included as disqualifying in nutritional indexes, could in some products not be linked with cardiovascular disease and total morbidity (Astrup et al., 2020). Therefore, more research is needed on if and how to include these negative and positive health aspects in nutritional metrics.

4. We recommend approaching the combination of environmental and nutritional aspects in a diet context in future studies, where possible. While this study focused on nutritional aspects in LCAs themselves, thus on a product level, diet level studies have the potential to combine environmental and nutritional factors in a more complete manner. Examples are diet optimization models (e.g., Broekema et al., (2020) and (Lucas et al., 2021)), in which environmental impacts of diets are minimized while ensuring nutritional adequacy.
5. Finally, we recommend to also consider aspects related to sustainable food systems beyond nutrition and environmental impact, including animal welfare, social, and economic impacts (Ridoutt, 2021b).

Conclusion

Based on the systematic review of recent studies described above, the following methods appear to be most appropriate to include in nutritional parameters in livestock LCA's on product level: i)

SAIN,LIM$_{5.3}$, ii) NRF$_{9.3}$, iii) NRF$_{11.3}$, and IV) NRF$_{5.3}$ (using same nutrients as in SAIN,LIM$_{5.3}$). These methods were selected because i) they are straightforward in terms of index calculation, ii) they allow to include multiple nutrients in one FU, iii) they assess the nutritional value of food products objectively and iv) they comply with most of the recommendations by the European Food Safety Authority regarding eligibility to health and nutrition claims. In future studies, these methods need to be further tested in a similar context to allow comparability.

References

Adriaanse, L.S. & Rensleigh, C. (2011). Comparing Web of Science, Scopus and Google Scholar from an Environmental Sciences perspective 1. South African Journal of Libraries and Information Science 77(2), https://doi.org/10.7553/77-2-58.

Astrup, A., Magkos, F., Bier, D.M., Brenna, J.T., de Oliveira Otto, M.C., Hill, J.O., King, J.C., Mente, A., . . . Krauss, R.M. (2020). Saturated Fats and Health: A Reassessment and Proposal for Food-Based Recommendations: JACC State-of-the-Art Review. Journal of the American College of Cardiology 76(7), 844-857. https://doi.org/10.1016/j.jacc.2020.05.077.

Behrens, P., Jong, J.K.-d., Bosker, T., Koning, A.D., Rodrigues, J.F.D. & Tukker, A. (2017). Evaluating the environmental impacts of dietary recommendations. Proceedings of the National Academy of Sciences 114(51), 13412-13417. https://doi.org/10.1073/pnas.1711889114.

Berardy, A., Johnston, C.S., Plukis, A., Vizcaino, M. & Wharton, C. (2019). Integrating Protein Quality and Quantity with Environmental Impacts in Life Cycle Assessment. Sustanability 11(10), 1-11. https://doi.org/10.3390/su11102747.

Bianchi, M., Strid, A., Winkvist, A., Lindroos, A.K., Sonesson, U. & Hallström, E. (2020). Systematic evaluation of nutrition indicators for use within food LCA studies. Sustainability 12(21), 1-18. https://doi.org/10.3390/su12218992.

Broekema, R., Tyszler, M., van 't Veer, P., Kok, F.J., Martin, A., Lluch, A. & Blonk, H.T.J. (2020). Future-proof and sustainable healthy diets based on current eating patterns in the Netherlands. The American Journal of Clinical Nutrition 112(5), 1338-1347. https://doi.org/10.1093/ajcn/nqaa217.

Bruno, M., Thomsen, M., Pulselli, F.M., Patrizi, N., Marini, M. & Caro, D. (2019). The carbon footprint of Danish diets. Climatic Change 156, 489-507.

D'Evoli, L., Salvatore, P., Lucarini, M., Nicoli, S., Aguzzi, A., Gabrielli, P. & Lombardi-Boccia, G. (2009). Nutritional value of traditional Italian meat-based dishes: Influence of cooking methods and recipe formulation. International Journal of Food Sciences and Nutrition 60(Suppl. 5), 38-49. https://doi.org/10.1080/09637480802322103.

Darmon, N., Vieux, F., Maillot, M., Volatier, J.L. & Martin, A. (2009). Nutrient profiles discriminate between foods according to their contribution to nutritionally adequate diets: A validation study using linear programming and the SAIN,LIM system. American Journal of Clinical Nutrition 89(4), https://doi.org/10.3945/ajcn.2008.26465.

Detzel, A., Krüger, M., Busch, M., Blanco-Gutiérrez, I., Varela, C., Manners, R., Bez, J. & Zannini, E. (2021). Life cycle assessment of animal-based foods and plant-based protein-rich alternatives: an environmental perspective. Journal of the Science of Food and Agriculture 102(12), https://doi.org/10.1002/jsfa.11417.

Doran-Browne, N.A., Eckard, R.J., Behrendt, R. & Kingwell, R.S. (2015). Nutrient density as a metric for comparing greenhouse gas emissions from food production. Climatic Change 129(1-2), 73-87. https://doi.org/10.1007/s10584-014-1316-8.

Drewnowski, A. (2009). Defining nutrient density: Development and validation of the nutrient rich foods index. Journal of the American College of Nutrition 28(4), https://doi.org/10.1080/07315724.2009.10718106.

EFSA (2008). The setting of nutrient profiles for foods bearing nutrition and health claims pursuant to Article 4 of the Regulation (EC) No 1924/2006 - Scientific Opinion of the Panel on Dietetic Products, Nutrition and Allergies. EFSA Journal 6(2), https://doi.org/10.2903/j.efsa.2008.644.

FAO/WHO/UNU (2007). Protein and amino acid requirements in human nutrition: report of a joint FAO/WHO/UNU expert consultation. Food and Agriculture Organization of the United Nations, World Health Organization & United Nations University. World Health Organization technical report series 935, https://apps.who.int/iris/handle/10665/43411.

Fresán, U., Mejia, M.A., Craig, W.J., Jaceldo-Siegl, K. & Sabaté, J. (2019). Meat analogs from different protein sources: A comparison of their sustainability and nutritional content. Sustainability 11(12), https://doi.org/10.3390/SU11123231.

Fulgoni, V.L., Keast, D.R. & Drewnowski, A. (2009). Development and validation of the nutrient-rich foods index: A tool to measure nutritional quality of foods. Journal of Nutrition 139(8), 1549-1554. https://doi.org/10.3945/jn.108.101360.

Gakidou, E., Afshin, A., Abajobir, A.A., Abate, K.H., Abbafati, C., Abbas, K.M., Abd-Allah, F., Abdulle, A.M., . . . Murray, C.J.L. (2017). Global, regional, and national comparative risk assessment of 84 behavioural, environmental and occupational, and metabolic risks or clusters of risks, 1990-2016: A systematic analysis for the Global Burden of Disease Study 2016. The Lancet 390, https://doi.org/10.1016/S0140-6736(17)32366-8.

Gaupp, F., Ruggeri Laderchi, C., Lotze-Campen, H., DeClerck, F., Bodirsky, B.L., Lowder, S., Popp, A., Kanbur, R., . . . Fan, S. (2021). Food system development pathways for healthy, nature-positive and inclusive food systems. Nature Food 2(12), 928-934. https://doi.org/10.1038/s43016-021-00421-7.

Godfray, H.C.J., Aveyard, P., Garnett, T., Hall, J.W., Key, T.J., Lorimer, J., Pierrehumbert, R.T., Scarborough, P., . . . Jebb, S.A. (2018). Meat consumption, health, and the environment. Science 243, https://doi.org/10.1126/science.aam5324.

Goglio, P., Knudsen, M.T., Van Mierlo, K., Röhrig, N., Fossey, M., Maresca, A., Hashemi, F., Waqas, M.A., . . . Smith, L.G. (2023). Defining common criteria for harmonizing life cycle assessments of livestock systems. Cleaner Production Letters 4, https://doi.org/10.1016/j.clpl.2023.100035.

Gómez, I., Janardhanan, R., Ibañez, F.C. & Beriain, M.J. (2020). The effects of processing and preservation technologies on meat quality: Sensory and nutritional aspects. Foods 9(10), https://doi.org/10.3390/foods9101416.

Green, A., Nemecek, T., Smetana, S. & Mathys, A. (2021). Reconciling regionally-explicit nutritional needs with environmental protection by means of nutritional life cycle assessment. Journal of Cleaner Production 312, https://doi.org/10.1016/j.jclepro.2021.127696.

Grossi, G., Goglio, P., Vitali, A. & Williams, A.G. (2019). Livestock and climate change: Impact of livestock on climate and mitigation strategies. Animal Frontiers 9(1), 69-76. https://doi.org/10.1093/af/vfy034.

Hauschild, M.Z., Rosenbaum, R.K. & Olsen, S.I. (2018). Life Cycle Assessment - Theory and Practice. Springer, Hitaj, C., Rehkamp, S., Canning, P. & Peters, C.J. (2019). Greenhouse Gas Emissions in the United States Food System: Current and Healthy Diet Scenarios. Environmental Science and Technology 53(9), 5493-5503. https://doi.org/10.1021/acs.est.8b06828.

ISO. (2006a). ISO 14040 Environmental management — Life cycle assessment — Principles and framework.

ISO. (2006b). ISO 14044 Environmental management — Life cycle assessment — Requirements and guidelines.

Jan, P., Repar, N., Nemecek, T. & Dux, D. (2019). Production intensity in dairy farming and its relationship with farm environmental performance : Empirical evidence from the Swiss alpine area. Livestock Science 224, 10-19. 10.1016/j.livsci.2019.03.019.

Jolliet, O. (2022). Integrating Dietary Impacts in Food Life Cycle Assessment. Frontiers in Nutrition 9, https://doi.org/10.3389/fnut.2022.898180.

Lucas, l., Guo, M. & Guillén-Gosálbez, G. (2021). Optimising diets to reach absolute planetary environmental sustainability through consumers. Sustainable Production and Consumption 28, 877-892. https://doi.org/10.1016/j.spc.2021.07.003.

Macombe, C., Loeillet, D. & Gillet, C. (2018). Extended community of peers and robustness of social LCA. International Journal of Life Cycle Assessment 23(3), https://doi.org/10.1007/s11367-016-1226-2.

Masset, G., Soler, L.G., Vieux, F. & Darmon, N. (2014). Identifying sustainable foods: The relationship between environmental impact, nutritional quality, and prices of foods representative of the french diet. Journal of the Academy of Nutrition and Dietetics 114(6), 862-869. https://doi.org/10.1016/j.jand.2014.02.002.

Masset, G., Vieux, F. & Darmon, N. (2015). Which functional unit to identify sustainable foods ? Public Health Nutrition 18(13), 2488-2497. https://doi.org/10.1017/S1368980015000579.

Mayer Labba, I.C., Steinhausen, H., Almius, L., Bach Knudsen, K.E. & Sandberg, A.S. (2022). Nutritional Composition and Estimated Iron and Zinc Bioavailability of Meat Substitutes Available on the Swedish Market. Nutrients 14(19), https://doi.org/10.3390/nu14193903.

McAuliffe, G.A., Chapman, D.V. & Sage, C.L. (2016). A thematic review of life cycle assessment (LCA) applied to pig production. Environmental Impact Assessment Review 56, 12-22. https://doi.org/10.1016/j.eiar.2015.08.008.

McAuliffe, G.A., Takahashi, T., Beal, T., Huppertz, T., Leroy, F., Buttriss, J., Collins, A.L., Drewnowski, A., . . . Lee, M.R.F. (2022). Protein quality as a complementary functional unit in life cycle assessment (LCA). International Journal of Life Cycle Assessment 28(2), 146-155. https://doi.org/10.1007/s11367-022-02123-z.

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 and Energy Security 7, e00143. https://doi.org/10.1002/fes3.143.

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. The International Journal of Life Cycle Assessment 25, 208-221. https://doi.org/10.1007/s11367-019-01679-7.

McLaren, S., Berardy, A., Henderson, A., Holden, N., Huppertz, T., Jolliet, O., De Camillis, C., Renouf, M., . . . Van Zanten, H. (2021). Integration of environment and nutrition in life cycle assessment of food items: opportunities and challenges. Rome, FAO, https://doi.org/10.4060/cb8054en.

McNicol, L.C., Perkins, L.S., Gibbons, J., Scollan, N.D., Nugent, A.P., Thomas, E.M., Swancott, E.L., McRoberts, C., White, A., Chambers, S. and Farmer, L., 2024. The nutritional value of meat should be considered when comparing the carbon footprint of lambs produced on different finishing diets. Frontiers in Sustainable Food Systems, 8, p.1321288.

Mehrabi, Z., Gill, M., van Wijk, M., Herrero, M. & Ramankutty, N. (2020). Livestock policy for sustainable development. Nature Food 1(3), https://doi.org/10.1038/s43016-020-0042-9.

Mogensen, L., Heusale, H., Sinkko, T., Poutanen, K., Sözer, N., Hermansen, J.E. & Knudsen, M.T. (2020). Potential to reduce GHG emissions and land use by substituting animal-based proteins by foods containing oat protein concentrate. Journal of Cleaner Production 274, https://doi.org/10.1016/j.jclepro.2020.122914.

Mullender, S.M., Sandor, M., Pisanelli, A., Kozyra, J., Borek, R., Ghaley, B.B., Gliga, A., von Oppenkowski, M., . . . Smith, L.G. (2020). A delphi-style approach for developing an integrated food/non-food system sustainability assessment tool. Environmental Impact Assessment Review 84, https://doi.org/10.1016/j.eiar.2020.106415.

Nassy, G., Girard, P. & Majou, D. (2021). Environmental labelling scheme test trials. Proposal for food-commodity environmental labelling built around appropriately-adapted functional units. Les Cahiers de l'IFIP 7(2),

Oonincx, D.G.A.B. & De Boer, I.J.M. (2012). Environmental Impact of the Production of Mealworms as a Protein Source for Humans - A Life Cycle Assessment. PLoS ONE 7(12), 1-5. https://doi.org/10.1371/journal.pone.0051145.

Poore, J. & Nemecek, T. (2018). Reducing food's environmental impacts through producers and consumers. Science 360(6392), 987-992. https://doi.org/10.1126/science.aaq0216.

Rayner, M., Scarborough, P., Heart, B. & Health, F. (2009). The UK Ofcom Nutrient Profiling Model. Defining 'healthy' and 'unhealthy' foods and drinks for TV advertising to children. UK Ofcom, https://www.ndph.ox.ac.uk/food-ncd/files/about/uk-ofcom-nutrient-profile-model.pdf.

Ridoutt, B. (2021a). An alternative nutrient rich food index (Nrf-ai) incorporating prevalence of inadequate and excessive nutrient intake. Foods 10(12), https://doi.org/10.3390/foods10123156.

Ridoutt, B. (2021b). Bringing nutrition and life cycle assessment together (nutritional LCA): opportunities and risks. The International Journal of Life Cycle Assessment 26, 1932-1936. https://doi.org/10.1007/s11367-021-01982-2.

RIVM. (2019). Dutch Nutrient Database (NEVO). National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands, https://www.rivm.nl/nederlands-voedingsstoffenbestand.

Röös, E., Carlsson, G., Ferawati, F., Hefni, M., Stephan, A., Tidåker, P. & Witthöft, C. (2020). Less meat, more legumes: Prospects and challenges in the transition toward sustainable diets in Sweden. Renewable Agriculture and Food Systems 35(2), 192-205.

Roux, P. (2021). Choix de l'Unité Fonctionnelle (UF) en ACV dans le secteur agro-alimentaire: Synthèse des principales références scientifiques. Montpellier.

Saarinen, M., Fogelholm, M., Tahvonen, R. & Kurppa, S. (2017). Taking nutrition into account within the life cycle assessment of food products. Journal of Cleaner Production 149, 828-844. https://doi.org/10.1016/j.jclepro.2017.02.062.

Salazar, T., Cai, H., Bailey, R., Huang, J.-y. & Bel, M. (2019). Defining nutritionally and environmentally healthy dietary choices of omega-3 fatty acids. Journal of Cleaner Production 228, https://doi.org/10.1016/j.jclepro.2019.04.359.

Schaubroeck, T., Ceuppens, S., Luong, A.D., Benetto, E., De Meester, S., Lachat, C. & Uyttendaele, M. (2018). A pragmatic framework to score and inform about the environmental sustainability and nutritional profile of canteen meals, a case study on a university canteen. Journal of Cleaner Production 187, https://doi.org/10.1016/j.jclepro.2018.03.265.

Schmitt, E., Keech, D., Maye, D., Barjolle, D. & Kirwan, J. (2016). Comparing the sustainability of local and global food chains: A case study of cheese products in Switzerland and the UK. Sustainability 8(5), https://doi.org/10.3390/su8050419.

Smith, N.W., Fletcher, A.J., Hill, J.P. & McNabb, W.C. (2022). Animal and plant-sourced nutrition: Complementary not competitive. Animal Production Science 62(8), 701-711. https://doi.org/10.1071/AN21235.

Springmann, M., Godfray, H.C.J., Rayner, M. & Scarborough, P. (2016). Analysis and valuation of the health and climate change cobenefits of dietary change. Proceedings of the National Academy of Sciences 113(15), 4146-4151. https://doi.org/10.1073/pnas.1523119113.

Springmann, M., Spajic, L., Clark, M.A., Poore, J., Herforth, A., Webb, P., Rayner, M. & Scarborough, P. (2020). The healthiness and sustainability of national and global food based dietary guidelines: Modelling study. The BMJ 370, 1-16. https://doi.org/10.1136/bmj.m2322.

Tello, A., Aganovic, K., Parniakov, O., Carter, A., Heinz, V. & Smetana, S. (2021). Product development and environmental impact of an insect-based milk alternative. Future Foods 4, https://doi.org/10.1016/j.fufo.2021.100080.

Tounian, P. (2022). Intérêt des produits carnés chez l'enfant et l'adolescent. Viandes & Produits Carnés, https://www.viandesetproduitscarnes.fr/index.php/fr/1219-interet-des-produits-carnes-chez-l-enfant-et-l-adolescent.

Tyszler, M., Kramer, G. & Blonk, H. (2016). Just eating healthier is not enough: studying the environmental impact of different diet scenarios for Dutch women (31–50 years old) by linear programming. International Journal of Life Cycle Assessment 21(5), 701-709. https://doi.org/10.1007/s11367-015-0981-9.

Van Dooren, C., Douma, A., Aiking, H. & Vellinga, P. (2017). Proposing a Novel Index Reflecting Both Climate Impact and Nutritional Impact of Food Products. Ecological Economics 131, 389-398. https://doi.org/10.1016/j.ecolecon.2016.08.029.

Van Mierlo, K., Rohmer, S. & Gerdessen, J.C. (2017). A model for composing meat replacers: Reducing the environmental impact of our food consumption pattern while retaining its nutritional value. Journal of Cleaner Production 165, 930-950. https://doi.org/10.1016/j.jclepro.2017.07.098.

Van Zanten, H.H.E., De Boer, I.J.M., Herrero, M., Van Hal, O., Elin, R., Muller, A., Garnett, T., Gerber, P.J. & Schader, C. (2018). Defining a land boundary for sustainable livestock consumption. Global Change Biology 24(9), 4185-4194. https://doi.org/10.1111/gcb.14321.

Vieux, F., Rémond, D., Peyraud, J.-L. & Darmon, N. (2022). Approximately Half of Total Protein Intake by Adults Must be Animal-Based to Meet Nonprotein, Nutrient-Based Recommendations, With Variations Due to Age and Sex. The Journal of Nutrition 152(11), 2514-2525. https://doi.org/10.1093/jn/nxac150.

Walker, C., Gibney, E.R., Mathers, J.C. & Hellweg, S. (2019). Comparing environmental and personal health impacts of individual food choices. Science of The Total Environment 685, 609-620. https://doi.org/10.1016/j.scitotenv.2019.05.404.

Weidema, B.P. & Stylianou, K.S. (2020). Nutrition in the life cycle assessment of foods-function or impact? International Journal of Life Cycle Assessment 25, 1210-1216. https://doi.org/10.1007/s11367-019-01658-y/Published.

Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., . . . Murray, C.J.L. (2019). Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. The Lancet Commissions, https://doi.org/10.1016/S0140-6736(18)31788-4.

Wu, G., Fanzo, J., Miller, D.D., Pingali, P., Post, M., Steiner, J.L. & Thalacker-Mercer, A.E. (2014). Production and supply of high-quality food protein for human consumption: Sustainability, challenges, and innovations. Annals of the New York Academy of Sciences 1321(1), 1-19. https://doi.org/10.1111/nyas.12500.

Xu, Z., Fu, Z., Zhai, Z., Yang, X., Meng, F., Feng, X., Zhong, J., Dai, Y., . . . Zhang, Z. (2020). Comparative evaluation of carbon footprints between rice and potato food considering the characteristic of Chinese diet. Journal of Cleaner Production 257, https://doi.org/10.1016/j.jclepro.2020.120463.

Xu, Z., Xu, W., Zhang, Z., Yang, Q. & Meng, F. (2017). Measurement and evaluation of carbon emission for different types of carbohydrate-rich foods in China. Chemical Engineering Transactions 61, 409-414. https://doi.org/10.3303/CET1761066.

SG5a: Harmonizing soil carbon simulation models, emission factors and direct measurements used in LCA of agricultural systems

Simone Pelaracci[a], Pietro Goglio[a], Simon Moakes[b,c], Marie Trydeman Knudsen[d,e], Klara Van Mierlo[f], Nina Adams[g], Fossey Maxime[h], Alberto Maresca[i], Manuel Romero-Huelva[j], Muhammad Ahmed Waqas[d,e], Laurence G. Smith[g, k], Frank Willem Oudshoorn[l], Thomas Nemecek[m], Camillo de Camillis[n], Giampiero Grossi[o], Ward Smith[p]

[a]Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy

[b]Department of Food System Sciences, Research Institute of Organic Agriculture (FiBL), Frick, Switzerland

[c]IBERS, Aberystwyth University, UK

[d]Department of Agroecology, Aarhus University, Blichers Allé 20, 8830 Tjele, Denmark

[e]Aarhus University Interdisciplinary Centre for climate change (iCLIMATE), Department of Agroecology, Blichers Alle 20, 8830 Tjele, Denmark

[f]Wageningen Social and Economic Research, Bronlan 103, 6708 WH, Wageningen, Netherlands

[g]School of Agriculture, Policy and Development, University of Reading, UK

[h]Institut de l'élevage (IDELE), 149 rue de Bercy, 75012 Paris, France

[i]SEGES Innovation P/S, Agro Food Park 15, 8200 Aarhus, Denmark

[j]Estación Experimental del Zaidín (CSIC), Profesor Albareda 1, 18008 Granada, Spain

[k]Department of Biosystems and Technology, Swedish University of Agricultural Sciences, Box 190, SE-234 22 Lomma, Sweden

[l]Innovation Centre for Organic farming, Agro Food Park 26, DK 8200 Aarhus, Denmark

[m]Agroscope, Life Cycle Assessment research group, CH-8046 Zurich, Switzerland

[n]Agrifood Systems and Food Safety (ESF) Division, Food and Agriculture Organization of the United Nations, Viale delle Terme di Caracalla, 00153 Rome, Italy

[o]Department of Agriculture and Forest Sciences, University of Tuscia-Viterbo, via San Camillo De Lellis, snc, 01100 Viterbo, Italy

[p]Agriculture and Agri-Food Canada, Science and Technology Branch, Ottawa, ON, Canada

Submitted to Agricultural Systems

Abstract

CONTEXT

The increasing demand for animal products, coupled with the need greenhouse gas (GHG) emissions from livestock production, highlights the urgency for effective mitigation strategies for livestock systems. Soil organic carbon (SOC) sequestration, a crucial approach for reducing atmospheric GHG concentrations, is often underrepresented in Life Cycle Assessments (LCA) of agricultural systems, largely due to methodological challenges in accurately accounting for soil carbon dynamics.

OBJECTIVE

To evaluate soil carbon simulation models, emission factors and direct measurements used in LCA towards developing a harmonized approach for including soil carbon sequestration in LCA. The goals were to: i) assess soil carbon simulation models, emissions factors and direct measurements used in LCAs of agricultural systems; ii) evaluate the strengths and weaknesses of these models; iii) provide recommendations for LCA practitioners; and iv) identify areas for future methodological improvements.

METHODS

A systematic review of soil carbon simulation models, emission factors and direct measurements used in LCA's of livestock systems was conducted, obtaining 263 relevant articles from an initial pool of 29,151. Fifteen soil carbon simulation models, three methods based on emission factors in addition to direct measurements were identified, and a modified Delphi participatory process categorized them into three tiers based on complexity and data requirements. Each method was evaluated against established criteria through expert workshops.

RESULTS AND CONCLUSIONS

The results showed an inverse relationship between applicability and accuracy of methods, making the choice of methodology critical to achieving high-quality LCA results. Recommendations emphasize selecting methods based on objectives and data availability, while being aware of the effect of the initial soil carbon level and the assessment time period when using soil carbon simulation models. In addition, this study identified current methodological challenges in assessing soil C dynamics in LCA of agricultural systems.

SIGNIFICANCE

This research provides a foundation for improving LCA practices and supports better decision-making in mitigating climate impacts of agricultural systems.

Keywords: LCA, cropping system, livestock system, soil CO_2 emissions

Introduction

Agriculture, forestry and other land use sectors contribute 22% of the total greenhouse gas (GHG) emissions which comprised 59 Gt of CO_2-eq in 2019, worldwide. Thus, all economic sectors should reduce GHG emissions, including agriculture (IPCC, 2022). Meanwhile, animal product demand is forecast to increase in the future due to the growing population and economic prosperity (Godfray et al., 2018). To compensate for this increase, there is a need for practices that reduce total atmospheric emissions (Kane and Solutions, 2015). Carbon sequestration, which is the removal and temporary storage of carbon from the atmosphere either in the permanent vegetation or soil, is seen as a potential pathway towards climate change mitigation. (Brandão et al., 2013; Don et al., 2024; Rodrigues et al., 2023). Soil organic carbon (SOC) is the main terrestrial carbon sink for reducing GHG emissions, with potential additional benefits, such as improving soil health, fertility, and agricultural production (Rodrigues et al., 2023; Wang et al., 2022). Soils constitute the largest pool of terrestrial organic C (~1,500 Pg C at 1 m depth; 2,400 Pg C at 2 m depth (Paustian et al., 2016)), which is three times the amount of CO_2 currently in the atmosphere (~830 Pg C) and 240 times current annual fossil fuel emissions (~10 Pg) (Batjes, 2014; Ciais et al., 2013; Lal et al., 2021; Le Quéré et al., 2016). Therefore, increasing net soil C storage by even a small percentage over a large area represents substantial C accumulation potential. Soil carbon dynamics approach an equilibrium depending *inter alia* on soil types, climate, and management practices. Management strategies can increase SOC content, but the soils ability to sequester carbon is constrained (Powlson et al., 2014).

Land management changes such as crop selection, switching from annual to perennial crops and vice versa, reduction of tillage, waste and residue management, and grazing practises, can contribute to SOC increase (Petersen et al., 2013); together with changes in land use which can be direct, if the change occurs within the production system being assessed, or indirect, if the change occurs as a consequence of production, but does not occur in the same place that caused the change (Planton, 2013; ISO, 2013). It is essential to adopt sustainable management practices and technological innovations to maximize carbon sequestration in the soil. Thus, land use change (LUC) and sustainable soil management are crucial for the effective sequestration of terrestrial organic carbon (Rodrigues et al., 2023).

CO_2 emissions from soils are evaluated mostly with regards to land management changes (e.g. tillage, fertilisation) (Pelaracci et al., 2022) and LUCs (from and to grassland/ cropland/ forest), following Intergovernmental Panel for Climate Change (IPCC) classification (McConkey et al., 2019; Ogle et al., 2019b, 2019a). Short-term biogenic carbon fluxes, such as occur within annual crops are not considered in GHG accounting. For example, during the night, vegetation acts as a carbon source through plant respiration, while decomposition crop residues in the soil release carbon into the atmosphere. In addition the yearly storage of carbon in agricultural products, by means of photosynthesis is not included, as products are used, and thereby oxidated to CO_2 within a few years.

We refer to soil CO_2 emissions as all emissions related to changes in carbon stock. Within livestock systems, when accounting for CO_2 flows in agro-ecosystems it is important to assess which management practices and changes in land-use can improve or mitigate the effects of climate change (Grossi et al., 2019; Jiang et al., 2023; Sykes et al., 2019).

Life Cycle assessment (LCA) can be used to assess environmental impacts of livestock systems and products. It has also been effective to assess land management practices and their impact on environmental performance of a cropping and grassland systems (Goglio et al., 2014; Rotz, 2018; Zaher et al., 2013). In order to improve the environmental assessments in the livestock systems, it is important to consider the interaction between cropping and livestock systems.

The importance of soil C sequestration and soil CO_2 is poorly reflected in current LCAs (Goglio et al., 2015; Petersen et al., 2013), since the majority of studies have not included soil C sequestration in the overall GHG estimations, mainly due to methodological limitations (Brandão et al., 2019). However, recently a few LCA studies have attempted to include soil C changes - using mainly modelling (Goglio et al., 2018; Jensen et al., 2024; Knudsen et al., 2019; Lefebvre et al., 2021; Petersen et al., 2013). Jensen et al. (2024) showed a 14% reduction in the carbon footprint of cabbage and Knudsen et al. (2019) showed a 5-18% reduction in carbon footprint of milk from different production systems due to inclusion of soil carbon changes in the LCA. Goglio et al. (2018) demonstrated through direct observation that soil carbon sequestration accounted for 62% of the total global warming potential (GWP) mitigation across the cropping systems and crops analysis. This highlights the significant role that soil carbon plays in the overall GHG budget of cropping systems and crops, underscoring the necessity of incorporating these factors into future LCA methodologies (Goglio et al., 2015; Paustian et al., 2016).

Furthermore, the use of satellite-based methods, such as remote sensing and spectral analysis, is emerging as a promising solution for assessing soil carbon content at larger scales, providing more detailed and continuous data on spatial variations in soil carbon. These methods, together with field measurements, can enhance the accuracy of carbon sequestration estimates in LCA models. (Morais et al., 2023; Pouladi et al., 2023)

In addition, there is an increasing need to assess livestock systems, taking into account present and future climate (Godfray et al., 2018; Willett et al., 2019). Improved LCA methodologies can capture systems' effects, crop-livestock interactions and circular economy aspects (Costa et al., 2020; Goglio et al., 2017; Grossi et al., 2019; Van Zanten et al., 2018) with a focus on C sequestration and GHG emissions (FIL-IDF, 2022; Goglio et al., 2023).

Grasslands play a crucial role in carbon sequestration, significantly contributing to GHG mitigation. Despite this, they remain understudied compared to croplands, even though they represent ~70% of global agricultural area (ITPS, 2015), which is 25% of the Earth's ice-free land surface (FAOSTAT, 2019), and store 28% to 37% of the terrestrial SOC pool (Paustian et al., 2016) which implies that they play a significant role in the global carbon and water cycles (Herrero et al., 2016; Wang and Fang, 2009). Most of the existing carbon tools were primarily developed for annual crops, and their ability to simulate SOC dynamics in grasslands is often limited (Ehrhardt et al., 2018). These ecosystems are particularly complex and difficult to investigate because of the wide range of management and environmental conditions they are exposed to (McSherry and Ritchie, 2013; Senapati et al., 2016; Soussana et al., 2010), leading to a large variability in their CO_2 source/sink capacity, such as the frequency and intensity of foliage removal and its fate (grazed on site or mowed and exported) (Herrero et al., 2016; Jérôme et al., 2014), difficulties in measuring soil productivity, spatial variability due to grazing and animal excreta (Dlamini et al., 2016; Oates and Jackson, 2014), and complexities in direct and accurate measurements of small changes in SOC stocks over short time periods in response to different management practices (Allen et al., 2010; Arrouays et al., 2012). Models like DNDC, DAYCENT, Century provide valuable tools for representing these processes, but their accuracy can still be enhanced due to the inherent complexities of grassland ecosystems. Therefore, advancing our understanding and improving the modelling of grasslands are essential for developing effective carbon management strategies that contribute to global sustainability goals.

Several harmonisation attempts for calculating GHG emissions were carried out in sectors other than agriculture (Segura-Salazar et al., 2019; Siegert et al., 2019), wines (Jourdaine et al., 2020), citrus fruit sector (Cabot et al., 2022) or food waste, proposing to better integrate between LCA and soil science (Morris et al., 2017) and for soil N_2O emissions in agricultural systems (Goglio et al., 2024). However, the integration and recommendation of harmonised estimation tools for livestock systems and a harmonization attempt for soil organic carbon change, has not been published, even though recent guidelines have been proposed by the Food and Agriculture Organisation (FAO, 2020, 2016a, 2016b, 2016c, 2016d, 2016e),

Within this study, we undertook a coherent harmonization approach for soil carbon simulation models, emission factors and direct measurements used in LCA with the objectives of: i) assessing soil carbon simulation models, emission factors and direct measurements used in LCA of agricultural systems; ii) evaluate strength and weaknesses of these estimation tools; iii) providing recommendations for LCA practitioners; iv) identifying the need for methodological improvements in future research.

Methodology

SCREENING AND REVIEW PROCEDURES

A systematic review of the existing literature was conducted to provide a comprehensive assessment on how LCA methodologies account for soil CO_2 in LCA of agricultural systems. To achieve this, a review protocol was developed (Figure 1), describing the search and screening process including an iterative process of article selection based on restrictive criteria.

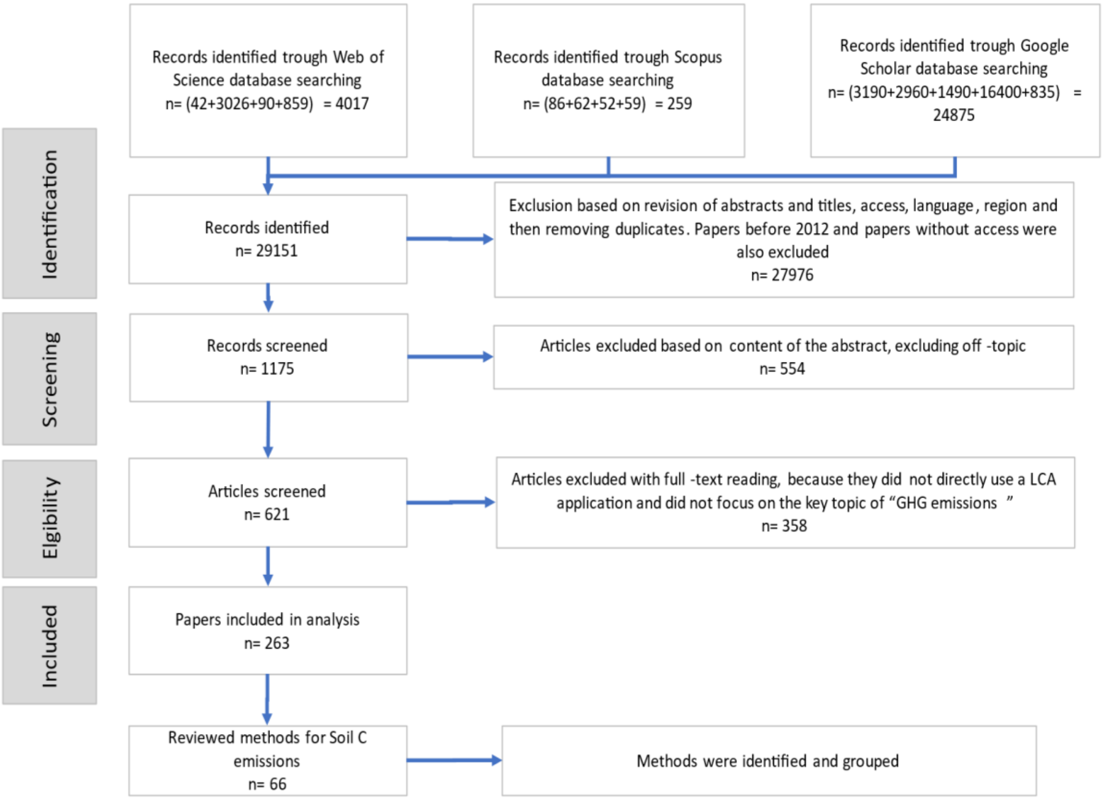
For the selection of scientific literature, publications in English in scientific journals or published by the FAO or the European Commission, were first retained.

A literature search was performed in Scopus, Web of Science and Google Scholar databases. Key words employed include "LCA", "Life Cycle Assessment"," life cycle analysis", "soil", "emissions", "carbon dioxide", "CO_2", "carbon sequestration", "GHG", "greenhouse gas", "C dynamics", "carbon", "livestock", "wheat", "maize", "grass", "barley", "oat", "soy", "faba beans", "alfalfa", "clover", "sorghum", "Rye", "Ley", "soil emissions", "soil carbon", "soil organic matter", "feed", "fodder", "farming system", "farm", "dairy", "cattle", "sheep", "pig", "poultry", "goat", "milk", "egg", "chicken", "cow", "husbandry", "crop soil emissions", "wheat soil emissions", and 29151 papers were found with these keywords. The search was limited to the 2012-2022 period in the following research areas: Agriculture; Agriculture or Soil or Animals or Cattle or Dairying or Crop production or Animal feed or Animal Husbandry or Swine or Livestock or Chickens or Poultry.

Selected publications focused on methods relevant to LCA that are linked to crop-livestock systems or their components, specifically applicable to crop-livestock systems. Papers related to rice, plastic, biofuel, and bioenergy were excluded as not fully related to the livestock sectors. Papers on biogas without any link to feed, insect, fish were also disregarded.

Further screening was carried out to analyse the evaluation of the accessibility of the articles, the language and the region. Documents prior to 2012 and inaccessible ones were excluded (1,175 documents). A further selection was based on the content of the abstracts, if relevant to the work. As a final step, the remaining 621 articles were subject to a complete reading of the text to exclude those not directly relevant. This iterative process brought the number of articles to 263, of which 66 related to soil C. After further grouping and method identification, only 20 were retained.

Figure 1 - Methodological steps of the literature search process for soil CO2 emissions



GENERAL CRITERIA AND SPECIFIC CRITERIA SELECTION

A harmonization participatory approach based on a modified Delphi method was used to identify key topics and evaluation criteria for LCAs of crop-livestock systems. The criteria were identified through a literature review and workshops (n=19) with experts from different disciplines and nationalities. These participatory approaches have fostered consensus among participants. The workshops were organized to elicit expert knowledge and record key findings, arguments and observations. Further details are provided in Goglio et al., (2023).

Initially, the priority topics on which to base the research were identified. An anonymous survey among LCA experts was conducted via Google Survey, to select the criteria, which were then refined through expert discussions to align with the methodological harmonization of LCAs for livestock systems and products. Definitions and scales have been adapted for some criteria to ensure rigor and consistency in the evaluation of LCA estimation tools.

The criteria that emerged from the discussion were: i) Transparency and reproducibility (Comprehensive documentation and mechanisms that allow reviewers to verify/review all data, calculations, and assumptions); ii) Completeness (Relationships between quantification of the environmental impact (material/energy flows and other environmental interventions) and adherence to the defined system boundary, the data requirements, and the impact assessment methods employed); iii) Fairness and acceptance (Level playing field across competing products, processes and industries); iv) Robustness (Associated in the RACER framework the following sub criteria of providing a defensible theory, Sensitivity, Data quality, Reliability, Consistency, Comparability, Boundaries); v) Applicability (Ability of the method to be used by a wide range of LCA practitioners).

The selection of specific criteria was carried out with a combined approach involving both literature and expert knowledge. A group of experts composed of three or four individuals, as in previous studies evaluating the implementation of LCA (Testa et al., 2022), was involved in the selection and refinement of the specific criteria (Goglio et al., 2023). The group worked on the specific assigned topic in three to five workshops. Four specific criteria related to soil C accounting estimation tools in crop-livestock systems were discussed: i) Adaptability to different soil types (If the method can be applied to different soil types, e.g. peat soils, coarse and medium/fine textured mineral soils); ii) Adaptability to different land uses (If the method can be applied to different types of land use, e.g. grassland and cropland); iii) Adaptability to different climates (If the method can be applied to different climates, e.g. temperate and boreal climates); iv) Accuracy (The ability of the LCA methods to capture the daily changes and the long-term dynamics of CO_2 emissions; it also takes into account the temporal horizon over which the soil CO_2 emissions occur (Brady and Weil, 2002; Lal and Stewart, 2018)).

It is assumed that the LCA practitioner has sufficient expertise to adopt the methodology and that observations have been carried out with a protocol. Further details can be found in Pelaracci et al. (2024).

DATA PROCESSING

Following the workshops with experts in which the general and specific criteria on which to evaluate the estimation tools were selected, a targeted discussion was held in which all the experts of subgroup 5 of the PATHWAYS project (15 experts from 12 research institutions and universities across Europe) evaluated all the estimation tools examined, using the chosen criteria.

Each expert evaluated each estimation tools for each criterion, assigning a score from 1 to 3 (or 1 to 4 based on the scale on which the criterion was evaluated). The overall method assessment was reviewed in several group workshops (n=19) which have been progressively evaluated. When

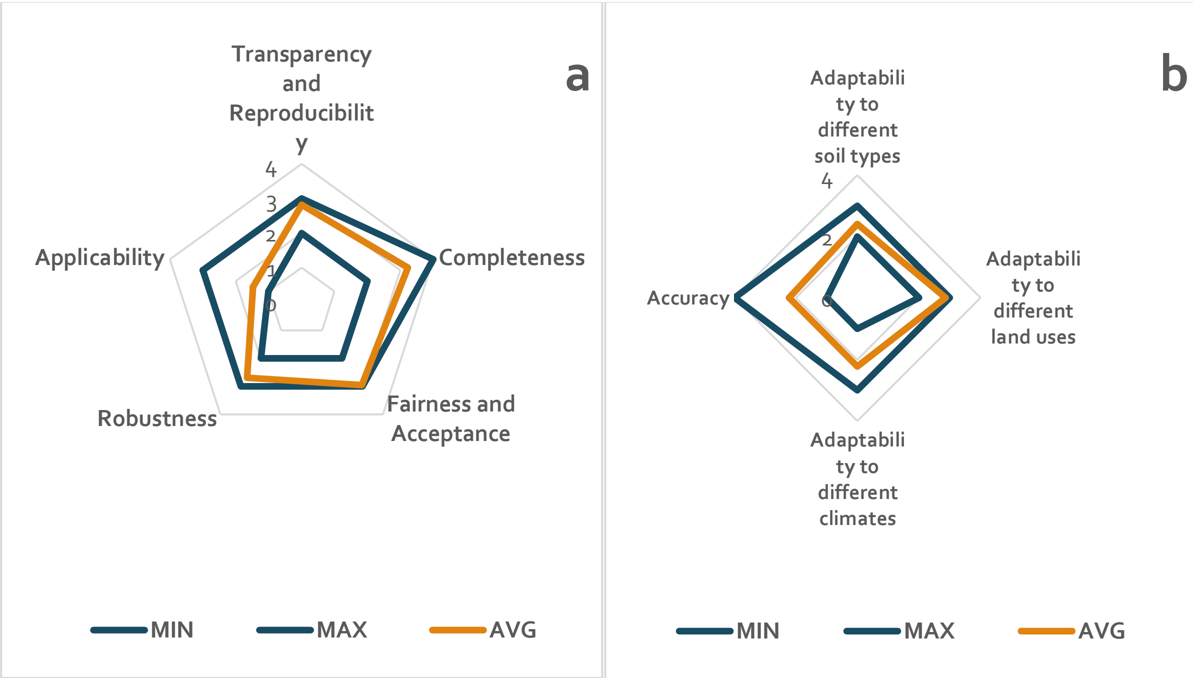
disagreement was found among experts, this was resolved through targeted discussions and reassessment of the methods, following previous research (Goglio et al., 2023).
From the data obtained, the mean, the minimum and maximum values were calculated for the scoring results for each different estimation tools and for each criterion (Fein et al., 2022).

Results

QUANTITATIVE RESULTS

The soil carbon simulation models, emission factors and direct measurements used in the LCA of livestock systems were acknowledged for most of the general criteria, with the exception of applicability (average score across all 5 criteria >2.68 on a scale of 1-3 except for completeness (1-4)). Average values were slightly higher than those found in the assessment for N_2O emission calculation methods in agricultural LCA (average score > 2.4) (Goglio et al., 2024). Despite higher average scores for general criteria, the applicability average score was lower (1.48 on average with a range from 1-3) (Figure 2) compared to N_2O emission methods (1.7 on average with a range from 1-3). More than 64% of soil CO_2 emissions reviewed in this research scored more than 3, indicating that the LCA estimation tools reviewed here have sufficient transparency, completeness, fairness, acceptance, and robustness, in contrast to Goglio et al. (2024) where 94% of the methods scored 2 or higher but only a smaller percentage (22%) scored 3 or higher. However, 55% of the estimation tools scored 1 for applicability, indicating that many estimation tools applied for soil carbon change have very limited applicability (Figure 2). Based on the estimation tools assessed, only the IPCC Tier I approach (for details see appendix B) scored 3 for applicability (Aalde et al., 2006).
For the specific criteria, the soil CO_2 emission estimation tools assessed had an average score above 2.23 on a 1-3 scale. However, for adaptability to soil types, land uses, and climate conditions, more than 96% of the estimation tools scored higher than 2. Except for adaptability to different climates, where the average scores were low (< 2.2 on a 1-3 scale). The methods for N_2O emissions also achieved high average scores (2.4 on a 1-3 scale) (Goglio et al., 2024). In contrast, only 18% scored above 2 for accuracy, with only three methods scoring 4 (i.e., CropSys, DNDC, and Delta LCA, see section appendix B) (Li et al., 1996; Stöckle et al., 2012; Wiedemann et al., 2016). Therefore, the majority of soil CO_2 emission estimation tools (82%) reviewed here were assessed as having low accuracy within livestock systems (Figure 2). This is similar to the findings in Goglio et al. (2024) for N_2O emission methods.
From the results obtained, approximately the same limitations for both soil carbon accounting estimation tools and N_2O emission methods were observed.

Figure 2 - Results obtained for the five general criteria (a) and four specific criteria (b) for the LCA methods used to assess soil CO_2 emissions models. Orange colour indicates the maximum value obtained, grey colour the minimum value and blue colour the average



MIN: Minimum value
MAX: Maximum value
AVG: Average value

IDENTIFIED KEY METHODOLOGICAL ISSUES

The soil carbon estimation tools, scored high with regards to the general criteria (>2.68 for all the general parameters except applicability). However, most of the estimation tools(59%) assessed have a low applicability (average value below 1.50). This can be related to the complexity and large data requirements of the estimation tools, limiting their applicability, as previously reported (Goglio et al., 2015). Most of the assessed soil carbon estimation tools (96%) considered climate, soil characteristics and land use, however only three estimation tools (DNDC, CropSys and Delta LCA) scored a high level of accuracy (>3), while the average for accuracy was quite low (<2). Furthermore Delta LCA can only be employed in Australian conditions (Wiedemann et al., 2016). All these estimation tools are based on several pools of carbon and are able to capture soil C dynamics (Li et al., 1996; Stöckle et al., 2012; Tuomi et al., 2009; Wiedemann et al., 2016). As highlighted in previous papers, it is often difficult to achieve high data quality for soil C assessments which is often the case for site-dependent LCA, site-generic LCA, consequential LCA and anticipatory LCA (Dale and Kim, 2014; Goglio et al., 2019; Potting and Hauschild, 2006). The most common estimation tools used to assess soil C is therefore the use of IPCC Tier 1 methodology, which has often been considered inadequate as it provides simplified estimates based on categories and poorly reflects local conditions, as previously reported (FAO, 2018; Goglio et al., 2015), which are relevant for LCA of agricultural systems (Camargo et al., 2013; MacWilliam et al., 2014). However this methodological compromise is highly dependent on the objectives and the system boundary of the assessment, in agreement with the ISO standards (ISO, 2006a, 2006b). The IPCC Tier 1 methodology scored very high in terms of applicability (3).

The carbon accounting estimation tools which scored higher in terms of accuracy, are often based on cropping systems and consider the field as one single average crop although there may be variation in yields within the field. That is, the model considers intercropping of multiple crops but only if they are similar crops as the average value between the two crops is used (Li et al., 1996; Stöckle et al., 2012; Tuomi et al., 2009; Wiedemann et al., 2016). The DNDC model, when used to simulate intercropping over the long-term (approximately 50 years), well simulated the yield and N uptake of the intercropping system under different N management scenarios, however , the yield and associated N uptake of one of the crops in the mix was underestimated (Zhang et al., 2018). On the other hand, grassland systems are multispecies systems, where each species has its own agronomic characteristics, which is often reflected in high spatial and temporal variability (Klumpp et al., 2010; Paustian et al., 2016). Furthermore, grassland yields and residues usually lack quantification at the farm level, making soil C dynamics more difficult to quantify through modelling (FAO, 2019).

Beside data quality and the type of methodology to be selected, another key factor is the LCA practitioner expertise. Independently of the method chosen, the inappropriate use of the soil CO_2 emission estimation tools could cause potential biases in the assessment, as previously discussed for soil C in agricultural LCA and for GHG mitigation (Goglio et al., 2015, 2019). A key aspect to be considered in the application of estimation tools for the assessment of soil carbon are the equilibrium dynamics of the soil C which affects the magnitude and duration of soil C sequestration (Paustian et al., 2016) going from one equilibrium reflected in the initial carbon content and depending on historical practices towards a new equilibrium based on the assessed farming practices. This equilibrium can be achieved with different timing and is dependent on the interaction between farm management, soil and climate characteristics (Gan et al., 2014; Goglio et al., 2015; Petersen et al., 2013). Indeed several models, such as DayCent or IPCC Tier 2 Steady State, require a spin-up period to stabilize the soil C dynamics (Pelletier et al., 2024; Uzoma et al., 2015). In DNDC a 5-10 years spin-up is required (He et al., 2021; Perlman et al., 2013). Thus, the initial soil carbon (reflecting historical practices) and the time perspective in which the assessments are done are indeed affecting the results of the soil CO_2 emission estimation tools.

Measurements if not appropriately carried out can also lead to biases (FAO, 2019). However, they are still a valuable data source for LCA, if properly carried out, despite their low applicability at a large scale due primarily to cost and time constraints (FAO, 2019; Goglio et al., 2018).

LCA METHODOLOGICAL ISSUES RELATED TO SCALE AND OBJECTIVES

The importance of soil C sequestration is poorly reflected in current LCA methodologies(Goglio et al., 2015; Koerber et al., 2009). Some LCA studies have included changes in soil carbon included changes in soil carbon based on a 100 years' time perspective to align with GWP100 (Knudsen et al., 2019, 2014) and other LCA studies have used temporal horizons of 30 years or less (Hörtenhuber et al., 2010; Röös et al., 2010; Halberg et al., 2010; Hillier et al., 2009; Mila i Canals et al., 2008; Gabrielle and Gagnaire, 2008), although the temporal horizon used is not explicitly stated in all studies. Most of the estimation tools discussed, do not fully consider the temporal effects of carbon balance in soil, which are relevant to climate change (Brandão et al., 2013, 2019; Bui et al., 2018; Plevin, 2017).

Among the main uncertainties and discussions regarding the inclusion of soil carbon changes in LCA of agricultural products is the achievement of a new equilibrium (Petersen et al., 2013). Essentially, the shift to a new agricultural practice will lead to a change towards a higher or lower level of soil organic matter, eventually stabilizing at a new equilibrium. The carbon in soil organic matter is not "stable" but undergoes constant turnover, and net changes in soil carbon will balance between what is sequestered and what is emitted (Oberholzer et al., 2014). Furthermore, some simple procedures such as Tier 1 IPCC use a 20-year temporal perspective to accumulate the total change in SOC between practices (time to equilibrium), however, the period for this to occur may actually be 30 years, or even 100 years (Goglio et al., 2015). As a result modelling only 10 or 20 years, the rate of accumulation of SOC, and thereby the consequences for GHG emission calculation, may be greatly exaggerated. Therefore, the chosen temporal perspective for assessing carbon sequestration or recovery time is crucial. A well verified process-based agroecosystem model can be used to estimate the period to equilibrium and also the dynamics of SOC change over time. Other approaches include using complex empirical models combined with a carbon decay model, such as the Bern Carbon Cycle Model, which allows the integration of temporal aspects of soil carbon changes by accounting for CO_2 degradation and atmospheric decline. This method highlights the significance of the time perspective chosen, with substantial differences observed across 20, 100, and 200-year horizons, thereby impacting the results and comparability in LCA applications (Petersen et al., 2013). Initially the rate of SOC change between practices is high with gradual decrease over time, usually following first order decay towards a new equilibrium (Smith et al., 2012).

Our assessment revealed that current carbon balance estimation tools, which have a significant impact on LCA results, show a dichotomy between high accuracy and low applicability, or vice versa (e.g., low accuracy but high applicability). Estimation tools with high applicability only roughly account for the interaction between soil, time, and management. Although the drivers of this interaction are well known, quantifying their effects on soil carbon is often difficult (Paustian et al., 2016), because of long-term equilibrium dynamics and soil variability (FAO, 2018; Loubet et al., 2011; Petersen et al., 2013). Most of the analysed empirical estimation tools consider constant management, while in reality, farmers may change crop management practices annually, thus influencing the outcomes on soil carbon dynamics (Goglio et al., 2017). However, the individual contributions of crop management practices to various carbon pools is usually not evaluated in the long term. Only a few attempts have been made, for example, using the Bern Carbon model (Petersen et al., 2013), the DAYCENT model (Nguyen et al., 2022) or DNDC model (Jiang et al., 2023).
The real challenge is how to include these estimates in climate impact assessment. Two main questions is the baseline (or initial soil carbon content depending on historical practices) and the time perspective of the assessments. In agricultural LCA's we often want to assess the impact of a certain agricultural practice. However, modelling the effect of this particular practice depends on this practice plus the initial soil carbon content. The initial soil carbon content is often decisive for the results of the modelling, and it depends on historical practices. If e.g. an arable crop rotation of grain legumes and catch crops are introduced on a soil with high soil C content due to a historical practice of dairy production, the soil C content will decrease - as opposed to if the same arable crop rotation was introduced to a soil with low C content due to a historical practice of intensive wheat production and straw removed, where the soil C content would increase. Thus, the interaction between the historical practice and the current practice of the land determines the LCA methods results, which is very important to be aware of when the focus in the LCA is on assessing effect of a current farming practice. Furthermore, the time perspective for the assessments can be decisive for the results since the time perspective from going from one equilibrium to another can be 20 to 100 years with the highest increase (or decrease) in the beginning. Thus, a short assessment period can be exaggerating the effect. These two main issues are very important to keep in mind when modelling the effects.
Future LCA research should therefore develop methodologies which encompass the correct level of details to capture the interaction between soil C dynamics and crop management on one side and on the other side the extensive application of the estimation tools in itself also by agricultural consultants and farmers with a more limited level of expertise. This methodological choice should be carried out in agreement with the LCA objectives.

LCA METHODOLOGICAL RECOMMENDATIONS

From the analysis of the current LCA estimation tools, some preliminary recommendations can be made regarding the suitability and application of estimation tools when undertaking an agricultural system LCA.

To accurately assess soil C dynamics within a temperate climate, a time perspective of at least 20 is required. This should be considered or, at the very least, estimated based on the best available knowledge (Goglio et al., 2015; Petersen et al., 2013), as such a "spin-up" period is necessary for most models. Furthermore, it is important to be aware of the shifts from one equilibrium to another and the potential decisive effect on the results of the historical practices reflected in the initial soil carbon content, when using estimation tools for accounting of C exchanges. For site-specific assessments (e.g., at the farm level), agroecosystem models such as DNDC or CropSys are preferred. If less detailed input data is available, the IPCC 2019 Tier 2 steady-state methodology can be employed. For broader, site-dependent or site-generic assessments, or when large-scale evaluations are needed, the use of Tier 2 methodologies such as the IPCC 2019 Tier 2 steady-state method or simplified carbon models like C-TOOL and ICBM is recommended (Andrén and Kätterer, 1997; Ogle et al., 2019; Petersen et al., 2013). In cases of very limited information, or when data quality cannot be ensured or expertise is lacking, the IPCC Tier 1 methodology may be used (Ogle et al., 2019). Regardless of the methodological approach chosen, it is essential to justify the choice and outline its potential limitations, in accordance with ISO standards (Alvarenga et al., 2012; ISO, 2006a, 2006b).

CONCLUSION

In this research an attempt to harmonize LCA estimation tools for agricultural systems was carried out together with providing recommendations for LCA practitioners and scientists. The identified estimation tools for GHG emissions focused on soil CO_2 emissions. Increasing net soil C storage by even a small percentage, represents substantial C accumulation potential and mitigation of GHG emissions, reducing climate impacts. It was observed that a high level of accuracy corresponded to a low level of applicability and vice versa. Thus, the choice of the methodology in relation to the LCA objectives is particularly critical to enable the best possible LCA assessments for the climate impact indicator.

Following the analysis of the available literature, a series of preliminary recommendations were proposed. As general recommendation for all the GHG from agricultural systems, the choice of LCA estimation tools for the individual impact categories should be based on the LCA objectives and data availability. For specifically the GHG assessments soil carbon balances are extremely important. More complex methods are available but they have greater data requirements and additional training or collaboration with modelling experts is required. Furthermore, it is crucial to be aware of the shift from one soil carbon equilibrium to the other and the potential decisive effect of the initial soil carbon content and the assessment period in time. At the other end of the complexity spectrum, the IPCC Tier 1 methodology has been employed in most of the assessments analysed here. Thus, for soil carbon there are only a few IPCC Tier 2 or basic process model solutions which combine the need for applicability with the need of accuracy. Independently of the estimation tool used, estimation tool limitations should be discussed in the LCA of agricultural systems.

The real challenge is how to include these estimates in climate impact assessment. Two main questions are the baseline (or initial soil carbon content depending on historical practices) and the time perspective of the assessments. The influence of past practices and crop types on the initial status of soil carbon will effect the results of the soil C accounting estimation tools. This problem may not exist when conducting a site-specific assessment for a single land unit or farm, as historical data may be available to make accurate estimates. However, when performing a site-dependent or site-generic large-scale assessment, such as evaluating soil carbon content at the national level, issues of overestimation or underestimation can arise due to the lack of historical data. This should be taken into account for future development of LCA methodology on soil carbon changes in agricultural systems. This LCA estimation tools development must be synchronous with improvements of modelling and observation methods and the assessment of different agricultural management practices.

FUNDING SOURCES

From This research has been developed within the PATHWAYS project, funded by the European Union's Horizon 2020 Research and Innovation Programme under grant agreement No 101000395.

References

Aalde, H., Gonzalez, Gytarsky, Krug, 2006. Vol 4: agriculture, forestry, and other land use, Chap. 2: generic methodologies applicable to multiple land-use., in: IPCC Guidelines for National Greenhouse Gas Inventories.

Allen, D., Pringle, M., Page, K., Dalal, R., 2010. A review of sampling designs for the measurement of soil organic carbon in Australian grazing lands. The Rangeland Journal 32, 227–246.

Alvarenga, R.A.F. de, da Silva Júnior, V.P., Soares, S.R., 2012. Comparison of the ecological footprint and a life cycle impact assessment method for a case study on Brazilian broiler feed production. Journal of Cleaner Production 28, 25–32. https://doi.org/10.1016/j.jclepro.2011.06.023

Andrén, O., Kätterer, T., 1997. ICBM: THE INTRODUCTORY CARBON BALANCE MODEL FOR EXPLORATION OF SOIL CARBON BALANCES. Ecological Applications 7, 1226–1236. https://doi.org/10.1890/1051-0761(1997)007[1226:ITICBM]2.0.CO;2

Arrouays, D., Marchant, B., Saby, N., Meersmans, J., Orton, T., Martin, M.P., Bellamy, P., Lark, R., Kibblewhite, M., 2012. Generic issues on broad-scale soil monitoring schemes: a review. Pedosphere 22, 456–469.

Batjes, N.H., 2014. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science 65, 10–21. https://doi.org/10.1111/ejss.12114_2

Brady, Weil, 2002. Brady, N., Weil, R., 2002. The Nature and Properties of Soils, 13th ed. Prentice Hall, Upper Saddle River, New Jersey, USA.

Brandão, M., Kirschbaum, M.U.F., Cowie, A.L., Hjuler, S.V., 2019. Quantifying the climate change effects of bioenergy systems: Comparison of 15 impact assessment methods. GCB Bioenergy 11, 727–743. https://doi.org/10.1111/gcbb.12593

Brandão, M., Levasseur, A., Kirschbaum, M.U.F., Weidema, B.P., Cowie, A.L., Jørgensen, S.V., Hauschild, M.Z., Pennington, D.W., Chomkhamsri, K., 2013. Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. Int J Life Cycle Assess 18, 230–240. https://doi.org/10.1007/s11367-012-0451-6

Bui, M., Adjiman, C.S., Bardow, A., Anthony, E.J., Boston, A., Brown, S., Fennell, P.S., Fuss, S., Galindo, A., Hackett, L.A., Hallett, J.P., Herzog, H.J., Jackson, G., Kemper, J., Krevor, S., Maitland, G.C., Matuszewski, M., Metcalfe, I.S., Petit, C., Puxty, G., Reimer, J., Reiner, D.M., Rubin, E.S., Scott, S.A., Shah, N., Smit, B., Trusler, J.P.M., Webley, P., Wilcox, J., Mac Dowell, N., 2018. Carbon capture and storage (CCS): the way forward. Energy Environ. Sci. 11, 1062–1176. https://doi.org/10.1039/C7EE02342A

Cabot, M.I., Lado, J., Clemente, G., Sanjuán, N., 2022. Towards harmonised and regionalised life cycle assessment of fruits: A review on citrus fruit. Sustainable Production and Consumption 33, 567–585.

Camargo, G.G.T., Ryan, M.R., Richard, T.L., 2013. Energy Use and Greenhouse Gas Emissions from Crop Production Using the Farm Energy Analysis Tool. BioScience 63, 263–273. https://doi.org/10.1525/bio.2013.63.4.6

Canals, L., Muñoz, I., Hospido, A., Plassmann, K., McLaren, S., Edwards-Jones, G., Hounsome, B., 2008. Life cycle assessment (LCA) of domestic vs. imported vegetables. Case Studies on Broccoli, Salad Crops and Green Beans.

Ciais, Sabine, Bala, Bopp, 2013. Ciais, P. et al. Carbon and other biogeochemical cycles. In Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (eds Stocker, T. F. et al.) 465–570 (Cambridge Univ. Press, 2013).

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. The International Journal of Life Cycle Assessment 25, 1942–1956. https://doi.org/10.1007/s11367-020-01812-x

Dale, B.E., Kim, S., 2014. Can the Predictions of Consequential Life Cycle Assessment Be Tested in the Real World? Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation..." Journal of Industrial Ecology 18, 466–467. https://doi.org/10.1111/jiec.12151

Dlamini, P., Chivenge, P., Chaplot, V., 2016. Overgrazing decreases soil organic carbon stocks the most under dry climates and low soil pH: A meta-analysis shows. Agriculture, Ecosystems & Environment 221, 258–269.

Don, A., Seidel, F., Leifeld, J., Kätterer, T., Martin, M., Pellerin, S., Emde, D., Seitz, D., Chenu, C., 2024. Carbon sequestration in soils and climate change mitigation—Definitions and pitfalls. Global Change Biology 30, e16983. https://doi.org/10.1111/gcb.16983

Ehrhardt, F., Soussana, J., Bellocchi, G., Grace, P., McAuliffe, R., Recous, S., Sándor, R., Smith, P., Snow, V., de Antoni Migliorati, M., 2018. Assessing uncertainties in crop and pasture ensemble model simulations of productivity and N2O emissions. Global Change Biology 24, e603–e616.

FAO, 2020. Livestock Environmental Assessment and Performance (LEAP) Partnership | Food and Agriculture Organization of the United Nations [WWW Document]. Food and Agriculture Organisation, Rome. URL http://www.fao.org/partnerships/leap/en/ (accessed 5.11.20).

FAO, 2019. Measuring and modelling soil carbon stocks and stock changes in livestock production systems: Guidelines for assessment (Version 1). Livestock Environmental Assessment and Performance (LEAP) Partnership. Rome, FAO. 170 pp. Licence: CC BY-NC-SA 3.0 IGO.

FAO, 2018. Measuring and modelling soil carbon stocks and stock changes in livestock production systems – Guidelines for assessment (Draft for public review). Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2016a. Environmental Performance of Pig Supply Chains: Guidelines for assessment (Livestock Environmental 251 Assessment and Performance Partnership). Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016b. Environmental performance of animal feeds supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016c. Greenhouse gas emissions and fossil energy use from small ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016d. Greenhouse gas emissions and fossil energy use from poultry supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016e. Environmental performance of large ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

Fein, E.C., Gilmour, J., Machin, T., 2022 Statistics for Research Students.

FIL-IDF, 2022. C-sequ. Life cycle assessment guidelines for calculating carbon sequestration in cattle production systems. Fédération Internationale du Lait-International Dairy Federation, Brussels.

Gabrielle, B., Gagnaire, N., 2008. Life-cycle assessment of straw use in bio-ethanol production: A case study based on biophysical modelling. Biomass and Bioenergy 32, 431–441. https://doi.org/10.1016/j.biombioe.2007.10.017

Gan, Y., Liang, C., Chai, Q., Lemke, R.L., Campbell, C.A., Zentner, R.P., 2014. Improving farming practices reduces the carbon footprint of spring wheat production. Nature Communications 5, 5012. https://doi.org/10.1038/ncomms6012

Godfray, H.C.J., Aveyard, P., Garnett, T., Hall, J.W., Key, T.J., Lorimer, J., Pierrehumbert, R.T., Scarborough, P., Springmann, M., Jebb, S.A., 2018. Meat consumption, health, and the environment. Science 361, eaam5324. https://doi.org/10.1126/science.aam5324

Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2017. Addressing crop interactions within cropping systems in LCA. Int. J. Life Cycle Assess. 1–9. https://doi.org/10.1007/s11367-017-1393-9

Goglio, P., Grant, B.B., Smith, W.N., Desjardins, R.L., Worth, D.E., Zentner, R., Malhi, S.S., 2014. Impact of management strategies on the global warming potential at the cropping system level. Science of The Total Environment 490, 921–933. https://doi.org/10.1016/j.scitotenv.2014.05.070

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. Cleaner Production Letters 4, 100035. https://doi.org/10.1016/j.clpl.2023.100035

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

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., Gao, X., Hanis, K., Tenuta, M., Campbell, C.A., McConkey, B.G., Nemecek, T., Burgess, P.J., Williams, A.G., 2018a. A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. J. Clean. Prod. 172. https://doi.org/10.1016/j.jclepro.2017.03.133

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J. Clean. Prod. 104, 23–39. https://doi.org/10.1016/j.jclepro.2015.05.040

Goglio, Pietro, Smith, W.N., Worth, D.E., Grant, B.B., Desjardins, R.L., Chen, W., Tenuta, M., McConkey, B.G., Williams, A., Burgess, P., 2018b. Development of Crop.LCA, an adaptable screening life cycle assessment tool for agricultural systems: A Canadian scenario assessment. Journal of Cleaner Production 172, 3770–3780. https://doi.org/10.1016/j.jclepro.2017.06.175

Goglio, P., Williams, A., Balta-Ozkan, N., Harris, N.R.P., Williamson, P., Huisingh, D., Zhang, Z., Tavoni, M., 2019. Advances and challenges of Life Cycle Assessment (LCA) of Greenhouse Gas Removal Technologies to Fight Climate Changes. J. Clean. Prod. 118896. https://doi.org/10.1016/j.jclepro.2019.118896

Grossi, G., Goglio, P., Vitali, A., Williams, A.G., 2019a. Livestock and climate change: impact of livestock on climate and mitigation strategies. Animal Frontiers 9, 69–76. https://doi.org/10.1093/af/vfy034

Halberg, N., Hermansen, J.E., Kristensen, I.S., Eriksen, J., Tvedegaard, N., Petersen, B.M., 2010. Impact of organic pig production systems on CO2 emission, C sequestration and nitrate pollution. Agronomy for Sustainable Development 30, 721–731. https://doi.org/10.1051/agro/2010006

Hauschild, M., 2006. Spatial Differentiation in Life Cycle Impact Assessment: A decade of method development to increase the environmental realism of LCIA. The International Journal of Life Cycle Assessment 11, 11–13. https://doi.org/10.1065/lca2006.04.005

He, W., Grant, B.B., Jing, Q., Lemke, R., St. Luce, M., Jiang, R., Qian, B., Campbell, C.A., VanderZaag, A., Zou, G., Smith, W.N., 2021. Measuring and modeling soil carbon sequestration under diverse cropping systems in the semiarid prairies of western Canada. Journal of Cleaner Production 328, 129614. https://doi.org/10.1016/j.jclepro.2021.129614

Herrero, M., Henderson, B., Havlík, P., Thornton, P.K., Conant, R.T., Smith, P., Wirsenius, S., Hristov, A.N., Gerber, P., Gill, M., 2016. Greenhouse gas mitigation potentials in the livestock sector. Nature Climate Change 6, 452–461.

Hillier, J., WHITTAKER, C., DAILEY, G., AYLOTT, M., CASELLA, E., RICHTER, G.M., RICHE, A., MURPHY, R., TAYLOR, G., SMITH, P., 2009. Greenhouse gas emissions from four bioenergy crops in England and Wales: Integrating spatial estimates of yield and soil carbon balance in life cycle analyses. GCB Bioenergy 1, 267–281. https://doi.org/10.1111/j.1757-1707.2009.01021.x

Hörtenhuber, S., Lindenthal, T., Amon, B., Markut, T., Kirner, L., Zollitsch, W., 2010. Greenhouse gas emissions from selected Austrian dairy production systems—model calculations considering the effects of land use change. Renewable Agriculture and Food Systems 25, 316–329. https://doi.org/10.1017/S1742170510000025

IPCC, 2022. Climate change 2022: Mitigation of climate change. WGIII Mitigation of Climate Change Climate Change 2022 Working Group III contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, Interngovernmental panel for climate change, Geneva, Switzerland.

ISO, 2006a. SS-EN ISO 14044 Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organization for Standardization, Geneva.

ISO, 2006b. SS-EN ISO 14040 Environmental Management- Life Cycle Assessment, Principles and Framework. International Organization for Standardization, Geneva.

ITPS, F. and, 2015. Status of the world's soil resources (SWSR)—Main report. Food and Agriculture Organization of the United Nations and intergovernmental technical panel on soils 650.

Jensen, A., Mogensen, L., van der Werf, H.M.G., Xie, Y., Kristensen, H.L., Knudsen, M.T., 2024. Environmental impacts and potential mitigation options for organic open-field vegetable production in Denmark assessed through life cycle assessment. Sustainable Production and Consumption 46, 132–145. https://doi.org/10.1016/j.spc.2024.02.008

Jérôme, E., Beckers, Y., Bodson, B., Heinesch, B., Moureaux, C., Aubinet, M., 2014. Impact of grazing on carbon dioxide exchanges in an intensively managed Belgian grassland. Agriculture, ecosystems & environment 194, 7–16.

Jiang, R., Jayasundara, S., Grant, B.B., Smith, W.N., Qian, B., Gillespie, A., Wagner-Riddle, C., 2023. Impacts of land use conversions on soil organic carbon in a warming-induced agricultural frontier in Northern Ontario, Canada under historical and future climate. Journal of Cleaner Production 404, 136902. https://doi.org/10.1016/j.jclepro.2023.136902

Jourdaine, M., Loubet, P., Trebucq, S., Sonnemann, G., 2020. A detailed quantitative comparison of the life cycle assessment of bottled wines using an original harmonization procedure. J. Clean. Prod. 250, 119472. https://doi.org/10.1016/j.jclepro.2019.119472

Kane, D., Solutions, L.L.C., 2015. Carbon sequestration potential on agricultural lands: a review of current science and available practices. National sustainable agriculture coalition breakthrough strategies and solutions, LLC 1–35.

Klumpp, K., Bloor, J.M.G., Ambus, P., Soussana, J.-F., 2010. Effects of clover density on N2O emissions and plant-soil N transfers in a fertilised upland pasture. Plant Soil 343, 97–107. https://doi.org/10.1007/s11104-010-0526-8

Knudsen, M.T., Dorca-Preda, T., Djomo, S.N., Peña, N., Padel, S., Smith, L.G., Zollitsch, W., Hörtenhuber, S., Hermansen, J.E., 2019. The importance of including soil carbon changes, ecotoxicity and biodiversity impacts in environmental life cycle assessments of organic and conventional milk in Western Europe. Journal of Cleaner Production 215, 433–443. https://doi.org/10.1016/j.jclepro.2018.12.273

Knudsen, M.T., Meyer-Aurich, A., Olesen, J.E., Chirinda, N., Hermansen, J.E., 2014. Carbon footprints of crops from organic and conventional arable crop rotations – using a life cycle assessment approach. Journal of Cleaner Production 64, 609–618. https://doi.org/10.1016/j.jclepro.2013.07.009

Koerber, G.R., Edwards-Jones, G., Hill, P.W., Canals, L.M. i, Nyeko, P., York, E.H., Jones, D.L., 2009. Geographical variation in carbon dioxide fluxes from soils in agro-ecosystems and its implications for life-cycle assessment. Journal of Applied Ecology 46, 306–314. https://doi.org/10.1111/j.1365-2664.2009.01622.x

Lal, R., Monger, C., Nave, L., Smith, P., 2021. The role of soil in regulation of climate. Philosophical Transactions of the Royal Society B: Biological Sciences 376, 20210084. https://doi.org/10.1098/rstb.2021.0084

Lal, R., Stewart, B.A. (Eds.), 2018. Soil and Climate, 1st ed. CRC Press, Boca Raton. doi: 10.1201/b21225

Le Quéré, C., Andrew, R.M., Canadell, J.G., Sitch, S., Korsbakken, J.I., Peters, G.P., Manning, A.C., Boden, T.A., Tans, P.P., Houghton, R.A., Keeling, R.F., Alin, S., Andrews, O.D., Anthoni, P., Barbero, L., Bopp, L., Chevallier, F., Chini, L.P., Ciais, P., Currie, K., Delire, C., Doney, S.C., Friedlingstein, P., Gkritzalis, T., Harris, I., Hauck, J., Haverd, V., Hoppema, M., Klein Goldewijk, K., Jain, A.K., Kato, E., Körtzinger, A., Landschützer, P., Lefèvre, N., Lenton, A., Lienert, S., Lombardozzi, D., Melton, J.R., Metzl, N., Millero, F., Monteiro, P.M.S., Munro, D.R., Nabel, J.E.M.S., Nakaoka, S., O'Brien, K., Olsen, A., Omar, A.M., Ono, T., Pierrot, D., Poulter, B., Rödenbeck, C., Salisbury, J., Schuster, U., Schwinger, J., Séférian, R., Skjelvan, I., Stocker, B.D., Sutton, A.J., Takahashi, T., Tian, H., Tilbrook, B., van der Laan-Luijkx, I.T., van der Werf, G.R., Viovy, N., Walker, A.P., Wiltshire, A.J., Zaehle, S., 2016. Global Carbon Budget 2016. Earth System Science Data 8, 605–649. https://doi.org/10.5194/essd-8-605-2016

Lefebvre, D., Williams, A., Kirk, G.J.D., Meersmans, J., Sohi, S., Goglio, P., Smith, P., 2021. An anticipatory life cycle assessment of the use of biochar from sugarcane residues as a greenhouse gas removal technology. Journal of Cleaner Production 312, 127764. https://doi.org/10.1016/j.jclepro.2021.127764

Li, C., Narayanan, V., Harriss, R.C., 1996. Model estimates of nitrous oxide emissions from agricultural lands in the United States. Global Biogeochem. Cycles 10, 297–306. https://doi.org/10.1029/96GB00470

Loubet, B., Laville, P., Lehuger, S., Larmanou, E., Fléchard, C., Mascher, N., Genermont, S., Roche, R., Ferrara, R.M., Stella, P., Personne, E., Durand, B., Decuq, C., Flura, D., Masson, S., Fanucci, O., Rampon, J.-N., Siemens, J., Kindler, R., Gabrielle, B., Schrumpf, M., Cellier, P.,

2011. Carbon, nitrogen and Greenhouse gases budgets over a four years crop rotation in northern France. Plant and Soil 343, 109–137. https://doi.org/10.1007/s11104-011-0751-9

MacWilliam, S., Wismer, M., Kulshreshtha, S., 2014. Life cycle and economic assessment of Western Canadian pulse systems: The inclusion of pulses in crop rotations. Agricultural Systems 123, 43–53. https://doi.org/10.1016/j.agsy.2013.08.009

McConkey, B., Ogle, S.M., Chirinda, N., Kishimoto-Mo, A.W., Baldock, J., Trunov, A., Alsaker, C., Lehmann, J., Woolf, D. (Eds.), 2019. Chapter 6: Grassland, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC, Intergovernmental panel for climate change, Geneva, Switzerland.

McSherry, M.E., Ritchie, M.E., 2013. Effects of grazing on grassland soil carbon: a global review. Global change biology 19, 1347–1357.

Morais, T.G., Jongen, M., Tufik, C., Rodrigues, N.R., Gama, I., Serrano, J., Gonçalves, M.C., Mano, R., Domingos, T., Teixeira, R.F.M., 2023. Satellite-based estimation of soil organic carbon in Portuguese grasslands. Frontiers in Environmental Science 11. https://doi.org/10.3389/fenvs.2023.1240106

Morris, J., Brown, S., Cotton, M., Matthews, H.S., 2017. Life-Cycle Assessment Harmonization and Soil Science Ranking Results on Food-Waste Management Methods. Environ. Sci. Technol. 51, 5360–5367. https://doi.org/10.1021/acs.est.6b06115

Nguyen, T.H., Field, J.L., Kwon, H., Hawkins, T.R., Paustian, K., Wang, M.Q., 2022. A multi-product landscape life-cycle assessment approach for evaluating local climate mitigation potential. Journal of Cleaner Production 354, 131691. https://doi.org/10.1016/j.jclepro.2022.131691

Oates, L.G., Jackson, R.D., 2014. Livestock management strategy affects net ecosystem carbon balance of subhumid pasture. Rangeland Ecology & Management 67, 19–29.

Oberholzer, H.R., Leifeld, J., Mayer, J., 2014. Changes in soil carbon and crop yield over 60 years in the Zurich Organic Fertilization Experiment, following land-use change from grassland to cropland. Journal of Plant Nutrition and Soil Science 177, 696–704. https://doi.org/10.1002/jpln.201300385

Ogle, S., Kurz, W.A., Green, C., Brandon, A., Baldock, J., Domke, J., Herold, M., Bernoux, M., Chirinda, N., De Ligt, R., Federici, S., Garcia, E., Grassi, G., Gschwantner, T., Hirata, Y., Houghton, R., House, J.J., Ishizuka, S., Jonckheere, I., Krisnawati, H., Lehtonen, A., Kinyanjui, M.J., McConkey, B., Naesset, E., Niinistö, S.M., Ometto, J.P., Panichelli, L., Paul, T., Peterson, H., Reddy, S., Regina, K., Rocha, M., Rock, J., Sanz-Sanchez, M., Sanquetta, S., Sato, S., Somogyi, Z., Trunov, A., Vazquez-Amabile, G., Vitullo, M., Wang, C., Waterworth, R.M., Collet, M., Harmon, M., Lehmann, J., Shaw, C.H., Shirato, Y., Wolf, D., 2019a. Chapter 2: Generic methodologies, applicable to multiple land-use categories., in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC, Intergovernmental panel for climate change, Geneva.

Ogle, S., Wakelin, S.J., Buendia, L., McConckey, B., Baldock, J., Akiyama, H., Kishimoto-Mo, A.M., Chirinda, N., Bernoux, M., Bhattacharya, S., Chuersuwan, N., Goheer, M.A.R., Hergoualc'h, K., Ishizuka, S., Lasco, R.D., Pan, X., Pathak, H., Regina, K., Sato, A., Vazquez-Amabile, G., Wang, C., Zheng, X., Alsaker, C., Cardinael, R., Corre, M.D., Gurung, R., Mori, A., Lehmann, J., Rossi, S., Van Straaten, O., Veldkamp, E., Woolf, d., Yagi, K., Yan, X., 2019b. Chapter 5: Cropland, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC International Panel on Climate Change, Geneva.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P., Smith, P., 2016. Climate-smart soils. Nature 532, 49–57. https://doi.org/10.1038/nature17174

[dataset] Pelaracci, S., 2024. General and specific criteria for evaluation of methods for accounting of C exchanges between soil and atmosphere in LCA of crop livestock-system. https://doi.org/10.5281/zenodo.10683622

Pelaracci, S., Rocchi, L., Romagnoli, F., Boggia, A., Paolotti, L., 2022. Agricultural Co-Product Management: An LCA Perspective on the Use of Safflower Oilcake from Bio-Oil Production in Umbria Region, Italy. Environmental and Climate Technologies 26, 25–35. https://doi.org/doi:10.2478/rtuect-2022-0003

Pelletier, THIAGARAJAN, Durnin-Vermette, Liang, Choo, Cerkowniak, Elkhoury, MacDonald, Smith, VandenBygaart, 2024. Bayesian Calibration of the Ipcc Tier-2 Steady-State Organic Carbon Model for Canadian Croplands Using Long-Term Experimental Data. https://dx.doi.org/10.2139/ssrn.4877052

Perlman, J., Hijmans, R.J., Horwath, W.R., 2013. Modelling agricultural nitrous oxide emissions for large regions. Environmental Modelling & Software 48, 183–192. https://doi.org/10.1016/j.envsoft.2013.07.002

Petersen, B.M., Knudsen, M.T., Hermansen, J.E., Halberg, N., 2013. An approach to include soil carbon changes in life cycle assessments. J. Clean. Prod. 52, 217–224. https://doi.org/10.1016/j.jclepro.2013.03.007

Planton, S., 2013. Annex III: glossary. Climate change 1447–1465.

Plevin, R.J., 2017. Assessing the Climate Effects of Biofuels Using Integrated Assessment Models, Part I: Methodological Considerations. Journal of Industrial Ecology 21, 1478–1487. https://doi.org/10.1111/jiec.12507

Potting, J., Hauschild, M.Z., 2006. Spatial differentiation in life cycle impact assessment: A decade of method development to increase the environmental realism of LCIA. Int. J. Life Cycle Assess. 11, 11–13. https://doi.org/10.1065/lca2006.04.005

Pouladi, N., Gholizadeh, A., Khosravi, V., Borůvka, L., 2023. Digital mapping of soil organic carbon using remote sensing data: A systematic review. CATENA 232, 107409. https://doi.org/10.1016/j.catena.2023.107409

Powlson, D.S., Stirling, C.M., Jat, M.L., Gerard, B.G., Palm, C.A., Sanchez, P.A., Cassman, K.G., 2014. Limited potential of no-till agriculture for climate change mitigation. Nature Climate Change 4, 678–683. https://doi.org/10.1038/nclimate2292

Rodrigues, C.I.D., Brito, L.M., Nunes, L.J.R., 2023. Soil Carbon Sequestration in the Context of Climate Change Mitigation: A Review. Soil Systems 7. https://doi.org/10.3390/soilsystems7030064

Röös, E., Sundberg, C., Hansson, P.-A., 2010. Uncertainties in the carbon footprint of food products: a case study on table potatoes. The International Journal of Life Cycle Assessment 15, 478–488.

Rotz, C.A., 2018. Modeling greenhouse gas emissions from dairy farms. Journal of Dairy Science 101, 6675–6690. https://doi.org/10.3168/jds.2017-13272

Segura-Salazar, J., Lima, F.M., Tavares, L.M., 2019. Life Cycle Assessment in the minerals industry: Current practice, harmonization efforts, and potential improvement through the integration with process simulation. J. Clean. Prod. 232, 174–192. https://doi.org/10.1016/j.jclepro.2019.05.318

Senapati, N., Jansson, P.-E., Smith, P., Chabbi, A., 2016. Modelling heat, water and carbon fluxes in mown grassland under multi-objective and multi-criteria constraints. Environmental Modelling & Software 80, 201–224.

Siegert, M.-W., Lehmann, A., Emara, Y., Finkbeiner, M., 2019. Harmonized rules for future LCAs on pharmaceutical products and processes. Int. J. Life Cycle Assess. 24, 1040–1057. https://doi.org/10.1007/s11367-018-1549-2

Smith, W.N., Grant, B.B., Campbell, C.A., McConkey, B.G., Desjardins, R.L., Kröbel, R., Malhi, S.S., 2012. Crop residue removal effects on soil carbon: Measured and inter-model comparisons. Agriculture, Ecosystems & Environment 161, 27–38. https://doi.org/10.1016/j.agee.2012.07.024

Soussana, J.-F., Tallec, T., Blanfort, V., 2010. Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. Animal 4, 334–350.

Stöckle, C., Higgins, S., Kemanian, A., Nelson, R., Huggins, D., Marcos, J., Collins, H., 2012. Carbon storage and nitrous oxide emissions of cropping systems in eastern Washington: A simulation study. Journal of Soil and Water Conservation 67, 365. https://doi.org/10.2489/jswc.67.5.365

Sykes, A.J., Topp, C.F.E., Rees, R.M., 2019. Understanding uncertainty in the carbon footprint of beef production. Journal of Cleaner Production 234, 423–435. https://doi.org/10.1016/j.jclepro.2019.06.171

Testa, F., Tessitore, S., Buttol, P., Iraldo, F., Cortesi, S., 2022. How to overcome barriers limiting LCA adoption? The role of a collaborative and multi-stakeholder approach. The International Journal of Life Cycle Assessment 27, 944–958.

Tuomi, M., Thum, T., Järvinen, H., Fronzek, S., Berg, B., Harmon, M., Trofymow, J.A., Sevanto, S., Liski, J., 2009. Leaf litter decomposition—Estimates of global variability based on Yasso07 model. Ecological Modelling 220, 3362–3371. https://doi.org/10.1016/j.ecolmodel.2009.05.016

Uzoma, K.C., Smith, W., Grant, B., Desjardins, R.L., Gao, X., Hanis, K., Tenuta, M., Goglio, P., Li, C., 2015. Assessing the effects of agricultural management on nitrous oxide emissions using flux measurements and the DNDC model. Agric. Ecosyst. Environ. 206, 71–83. https://doi.org/10.1016/j.agee.2015.03.014

Van Zanten, H.H.E., Herrero, M., Van Hal, O., Röös, E., Muller, A., Garnett, T., Gerber, P.J., Schader, C., De Boer, I.J.M., 2018. Defining a land boundary for sustainable livestock consumption. Glob. Change Biol. 24, 4185–4194. https://doi.org/10.1111/gcb.14321

Wang, W., Fang, J., 2009. Soil respiration and human effects on global grasslands. Global and Planetary Change 67, 20–28.

Wang, Y., Tao, F., Chen, Y., Yin, L., 2022. Interactive impacts of climate change and agricultural management on soil organic carbon sequestration potential of cropland in China over the coming decades. Science of The Total Environment 817, 153018. https://doi.org/10.1016/j.scitotenv.2022.153018

Wiedemann, S.G., Yan, M.-J., Henry, B.K., Murphy, C.M., 2016. Resource use and greenhouse gas emissions from three wool production regions in Australia. Journal of Cleaner Production 122, 121–132. https://doi.org/10.1016/j.jclepro.2016.02.025

Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., 2019. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. The lancet 393, 447–492.

Zaher, U., Stöckle, C., Painter, K., Higgins, S., 2013. Life cycle assessment of the potential carbon credit from no- and reduced-tillage winter wheat-based cropping systems in Eastern Washington State. Agricultural Systems 122, 73–78. https://doi.org/10.1016/j.agsy.2013.08.004

Zhang, Y., LIU, J., WANG, H., LEI, Q., LIU, H., ZHAI, L., REN, T., ZHANG, J., 2018. Suitability of the DNDC model to simulate yield production and nitrogen uptake for maize and soybean intercropping in the North China Plain. Journal of Integrative Agriculture 17, 2790–2801. https://doi.org/10.1016/S2095-3119(18)61945-8

SG5b: Harmonizing methods to account for soil nitrous oxide emissions in Life Cycle Assessment of agricultural systems

Pietro Goglio[a], Simon Moakes[b,c], Marie Trydeman Knudsen[d], Klara Van Mierlo[e], Nina Röhrig[f], Fossey Maxime[g], Alberto Maresca[h], Manuel Romero-Huelva[i], Muhammad Ahmed Waqas[d], Laurence G. Smith[f, j], Giampiero Grossi[k], Ward Smith[l], Camillo De Camillis[m], Thomas Nemecek[n], Francesco Tei[a], Frank Willem Oudshoorn[o]

[a]Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy

[b]Department of Food System Sciences, Research Institute of Organic Agriculture (FiBL), Frick, Switzerland

[c]IBERS, Aberystwyth University, UK

[d] Department of Agroecology, Aarhus University, Blichers Allé 20, 8830 Tjele, Denmark

[e]Wageningen Social and Economic Research, Droevendaalsesteeg 4, 6708 PB, Wageningen, The Netherlands

[f]School of Agriculture, Policy and Development, University of Reading, UK

[g]Institut de l'élevage (IDELE), 149 rue de Bercy, 75012 Paris, France

[h]SEGES Innovation P/S, Agro Food Park 15, 8200 Aarhus, Denmark

[i]Estación Experimental del Zaidín (CSIC), Profesor Albareda 1, 18008 Granada, Spain

[j]Department of Biosystems and Technology, Swedish University of Agricultural Sciences, Box 190, SE-234 22 Lomma, Sweden

[k]Department of Agriculture and Forests Sciences, University of Tuscia-Viterbo, via San Camillo De Lellis 01100 Viterbo, Italy.

[l]Agriculture and Agri-Food Canada, Ottawa Research and Development Centre, Ottawa, ON, Canada

[m]Food and Agriculture Organization of the United Nations (FAO), via delle Terme di Caracalla, 00153 Roma, Italy

[n]Agroscope, Life Cycle Assessment research group, 8046 Zurich, Switzerland

[o]Innovation Centre for Organic farming, Agro Food Park 26, DK 8200 Aarhus, Denmark

SG5b report was published in Agricultural Systems. Reference to the publication:
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. Agr. Syst. 219, 104015.
https://doi.org/10.1016/j.agsy.2024.104015

Abstract

Context

Worldwide greenhouse gas emissions (GHG) reached 59 Gt of CO_2eq in 2019 and agricultural soils are the primary source of N_2O emissions. Life cycle assessments (LCA) have been successful in assessing GHG from agricultural systems. However, no review and harmonization attempt has been focused on soil N_2O emissions, despite the need to improve LCA methodologies for assessing GHG in agricultural LCA.

Objective

We therefore undertook a review and harmonization of existing methods to account for soil N_2O emissions in LCA of agricultural systems and products: i) to compare current methods used in LCA; ii) to identify advantages and iii) disadvantages of each method in LCA; iv) to suggest recommendations for LCA of agricultural systems; v) to identify research needs and potential methodological developments to account for soil N_2O emissions in the LCA of agricultural systems. In this paper, we consider as soil N_2O emissions, those originated from soils in relation to fertilisers (organic and manufactured), crop residues, land use/land management change, grassland management, manure and slurry applications and from grazing animals.

Methods

The approach adopted was based on two anonymous expert surveys and a series of expert workshops (n=21) to define general and specific criteria to review LCA methods for GHG emissions used in LCA of agricultural systems. A broad list of keywords and search criteria was used as the research involved GHG assessment in agricultural LCA. Reviewed papers and methodology were then assessed by LCA and soil N_2O emission experts (n=14).

Results and discussion

More than 25000 scientific papers and reports were identified, 1175 were screened, 263 included in the final review and 31 scientific papers were related to soil N_2O emissions. The results showed that a high level of accuracy corresponded to a low level of applicability and vice versa, following the assessment framework developed in this work through participatory approaches.

Significance

The choice of LCA methods, critical for high quality LCA of agricultural systems, should be based on the assessment objectives, data availability and expertise of the LCA practitioner. However, it is preferable to use DNDC model after calibration and validation or direct field measurements, considering system effects. When necessary data are lacking, IPCC tier 2 methodology where available should be used, otherwise 2019 IPCC Tier 1 methodology. This LCA method development should be synchronous with improvements of quantification methods and the assessment of a wider range of agricultural management practices and systems.

Keywords: LCA, cropping systems, livestock systems, soil N_2O emissions, methods, harmonization

Introduction

Worldwide greenhouse gas (GHG) emissions reached 59 Gt of CO_2eq in 2019, while N_2O represents 4% of the total global emissions. However, following the Intergovernmental Panel for Climate Change 6th assessment report, nitrous oxide has a global warming potential with a 100 year horizon 273 times larger than carbon dioxide. Agriculture, forestry and land use sector contributed 22% of the total global GHG emissions (IPCC, 2022). Because of the large amounts of GHG emissions, there is an increasing demand for GHG emission reduction for every sector of the economy, including agriculture (IPCC, 2022).
Agricultural soils are the primary source of anthropogenic nitrous oxide (N_2O) emissions (Wang et al., 2018). Soil emissions due to synthetic fertilizer applications to soils accounted for 0.75% of the total global GHG emissions in 2019 (IPCC, 2022). N_2O emissions from manure management contributed 5% to global greenhouse gas emissions within the livestock production chains, while feed production accounted for 9.8% in 2015 (FAO, 2023).
Soil N_2O emissions are by-products of microbial processes transforming nitrate to nitrogen gas under microaerobic and anaerobic conditions (denitrification) or ammonium to nitrate under aerobic conditions (nitrification) (Oertel et al., 2016; Ussiri and Lal, 2013). These emissions are part of the N nitrogen cycle together with other pollutants (e.g. Ammonia, nitrate) which can cause other impacts such as acidification and eutrophication (Brady and Weil, 2002).
Nitrous oxide emissions are largely affected by the soil moisture and soil oxygen availability making these emissions highly variable throughout the season (Bastos et al., 2021; Dorich et al., 2020; Olesen et al., 2023). Indeed, a key parameter is the water filled pore space (WFPS), WFPS value above 60% creates favourable conditions for soil N_2O emissions through denitrification (Laville et al., 2011), but optimum N_2O production may occur at about 80% WFPS (Butterbach-Bahl et al., 2013). Thus, climate and soil types affect soil N_2O emissions (Butterbach-Bahl et al., 2013; Dorich et al., 2020; Loubet et al., 2011).
Further, N fertilizer use, N content and C/N ratio of manure or slurry, and the C/N ratio of crop residues also influence soil N_2O emissions (Dorich et al., 2020; Kimming et al., 2011; Saggar, 2010; Tuomisto et al., 2012; Ussiri et al., 2009). Soil N_2O emissions are often characterized by peak emission

events after fertilizer or manure applications, freezing-thaw periods and ploughing of grass, where most of the emissions occur during the growing season (Dorich et al., 2020; Giltrap et al., 2020; Olesen et al., 2023; Taki et al., 2019). Otherwise, the background N_2O emissions are generally low in concentration which makes field monitoring difficult and costly (Goglio et al., 2013; Laville et al., 2011; Olesen et al., 2023).

Accounting for fluxes of N_2O in LCA of agro-ecosystems is important for evaluating which management practices may enhance or mitigate climate change effects for different crop-livestock systems (Grossi et al., 2019; Sykes et al., 2019). Soil N_2O emissions from soils are evaluated mostly with regards to land management and land management changes (e.g. tillage, fertilizer application), and land use changes (from and to grassland/ cropland/ forest), following intergovernmental panel for climate change (IPCC) classification (McConkey et al., 2019; Ogle et al., 2019a, 2019b).

Life Cycle assessment (LCA) is an assessment method commonly used to assess crop, livestock systems and products due to its ability to identify environmental hotspots and trade-offs across different types of pollution (Cederberg et al., 2013), use of resources (e.g. energy and materials), biodiversity and human health impacts (Huijbregts et al., 2017; van der Werf et al., 2020; Zampori and Pant, 2019). LCA has also been widely used to assess climate change impacts of agricultural products and production systems (Grossi et al., 2019; Poore and Nemecek, 2018). This includes the assessment of different types of fertilizer, tillage practices and residues management within cropping systems (Goglio et al., 2014; Nemecek et al., 2015; Zaher et al., 2013). Other LCA research assessed the influence of the method used to estimate N_2O emissions on the overall LCA results of agricultural systems (Cabot et al., 2023; Goglio et al., 2018; Sinisterra-Solís et al., 2020).

Recently, a combined approach has been proposed for assessing livestock products and systems taking into account crop-livestock interaction (Ershadi et al., 2020; Marton et al., 2016; Parajuli et al., 2018). Considering the importance of mitigating GHG emissions there is an increasing need to assess complex livestock systems under current and future climate (Godfray et al., 2018; Willett et al., 2019). Furthermore, improved LCA methodologies are required to better capture systems effects, crop-livestock interactions and circular economy (Costa et al., 2020; Grossi et al., 2019; Van Zanten et al., 2018).

Several harmonisation attempts were focused mostly on sectors other than agriculture (Segura-Salazar et al., 2019; Siegert et al., 2019; UNEP, 2023a, 2023b), while others specifically focused on wines (Jourdaine et al., 2020), food waste advocating for a better integration between LCA and soil science (Morris et al., 2017) or generally on livestock systems (FAO, 2020). No harmonization attempt exists for soil N2O emissions in the LCA of agricultural systems, including crop-livestock interaction (FAO, 2020). Within this study, we therefore undertook a review and harmonization of existing methods to account for soil N2O emissions in Life Cycle Assessment in agricultural systems, including excreted N on pasture, and products: i) to compare current methods used in LCA; ii) to identify advantages and iii) disadvantages of each method in LCA; iv) to suggest recommendations for LCA of crop-livestock systems; v) to identify research needs and potential methodological developments to account for soil N2O emissions in the LCA of agricultural systems. In this paper, we consider soil N2O emissions as those originated from soils in relation to fertilisers (organic and manufactured), crop residues, land use/land management change, grassland management, manure and slurry applications and from grazing animals. All the manure management emissions related to manure handling, storage and animal housing are out of scope of the present research as they do not originate from soil. This paper is part of a broader research project (PATHWAYS) aiming at assessing pathways to sustainability for livestock and food systems integrating crop-livestock interactions. In particular, the research presented in this paper is part of an effort to harmonize LCA methods related to GHG emissions in LCA of crop-livestock systems and soil N_2O emissions were investigated together with soil C, manure emissions and enteric fermentation. However, this paper will only present and discuss the outcomes limited to soil N_2O emissions.

Methodology

SEARCH CRITERIA

A systematic literature search was conducted using Scopus, Google Scholar and the Web of science search engines. The systematic literature search and review had a broader scope, which was to identify methodologies to assess soil C sequestration, soil N_2O emissions and enteric fermentation; rather than just soil N_2O emissions in the LCA of agricultural systems, as described by Goglio et al., (2023a). Thus, search terms and search term combinations employed are described below in Table 1, including all papers published between 2012 to 2022. These were selected as considered relevant for LCA of crop-livestock and agricultural systems and for soil N_2O emissions.

SCREENING AND REVIEW PROCEDURES

The collected sources were screened against the following criteria: i) Peer-reviewed publications in a scientific journal, published by the European Commission, FAO or other international organizations; ii) English language publication; iii) Method is related to and applicable for LCA; iv) Method is related to agricultural systems or their components; v) Method is applicable for agricultural systems. A systematic review of the existing literature, based on the methodology described above, was conducted to provide a comprehensive assessment on how LCA methodologies include livestock GHG emissions in relation to soil N_2O from both cropland and grassland within crop-livestock systems. These include cropping systems receiving manure, sludge, slurry, grazed systems and all the related crop and grassland management practices (e.g. tillage, fertilizer management, residue management, weed control, irrigation). To achieve this, a review protocol was developed (Figure 1), describing the search and screening process including an iterative process of article selection based on restrictive criteria.

First ("*identification step*"), the literature search was performed, according to the queries defined in Table 1, in Scopus, Web of Science and Google Scholar databases. Searches led to a total of 29 151 papers. When the Google search engine was used in the search, the selection of papers was stopped at page 15 of the search results (Each Google Scholar page contained approximately 10 items). Papers with research which was not fully relevant to the crop-livestock sector such as rice, plastic, biofuel, and bioenergy were excluded. Energy papers related to biogas without any relation to feed, and soil emissions were also excluded as were papers with insects, fish or feed production without any focus on livestock.

Table 1: Combinations of search terms for the subgroup "GHG Emission Issues"

Database	Combination	Search strings[a]
Scopus & Web of Science	1	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("enteric fermentation")
	2	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("soil*") AND ("emissions" OR "nitrous oxide" OR "N2O" OR "carbon dioxide" OR "CO2" OR "carbon sequestration" OR "GHG" OR "greenhouse gas*" OR "C dynamics" OR "soil) AND ("carbon") AND ("livestock")
	3	("Life Cycle Assessment" OR " life cycle analysis") AND ("wheat" OR "maize" OR "grass" OR "barley" OR "oat" OR "soy*" OR "faba beans" OR "alfalfa" OR "clover" OR "sorghum" OR "Rye" OR "Ley") AND ("soil emissions" OR "soil carbon" OR "soil nitrogen" OR "soil organic matter" OR "nitrous oxide") AND ("feed" OR "fodder" OR "farming system" OR "farm")
	4	("Life Cycle Assessment" OR " life cycle analysis") AND ("livestock" OR "dairy" OR "cattle" OR "sheep" OR "pig*" OR "poultry" OR "goat*" OR "milk" OR "egg*" OR "chicken*" OR "cow*" OR "husbandry") AND ("emissions") NOT ("waste" OR "biofuel" OR "bioenergy")
	5	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("manure" OR "slurry") AND ("handling" OR "storage" OR "treatment" OR "emissions")
	6	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("emissions") AND ("livestock*" OR "dairy" OR "sheep" OR "pig" OR "poultry" OR "goat" OR "milk" OR "egg*" OR "Chicken" OR "cow "NOT "waste" OR "biofuel" OR "bioenergy")
	7	"LCA" "enteric fermentation" OR "enteric emissions"

Google Scholar	8	"LCA" "manure application" OR "manure emissions"
	10	"LCA" "crop soil emissions"
	11	"LCA" "livestock"
	18	"LCA" "wheat soil emissions"

[a]Last access in March 2022

The second step involved the review of abstracts and titles, article accessibility, language, region and removal of duplicate papers. The "screening" was accomplished by using restrictive criteria ("refine results") excluding appearances before 2012 and papers which were not accessible (1175 papers). Further selection was performed based on the content of the abstract and by excluding off-topic material. Finally, 621 papers were selected as "Eligible" for full-text reading.

After the full-texts were read, the final step was to exclude papers which were not directly used in LCA application or did not focus on the key topic of "GHG emissions". This resulted in a 263 papers included in the qualitative analysis related to soil C, soil N_2O emissions, manure emissions and enteric fermentation. Of these, 31 papers dealt with soil N_2O emissions in LCA of agriculture systems and 16 were identified as describing key methods *"Method identification"*. Direct measurements methods have been added in this assessment even if they are not part of a LCA of crop-livestock systems, as they have been reported in major publications related to greenhouse gas emissions, such as the IPCC (De Klein et al., 2006; Hergoualc'h et al., 2019), or used in LCA of cropping systems (Goglio et al., 2018; Zaher et al., 2013).

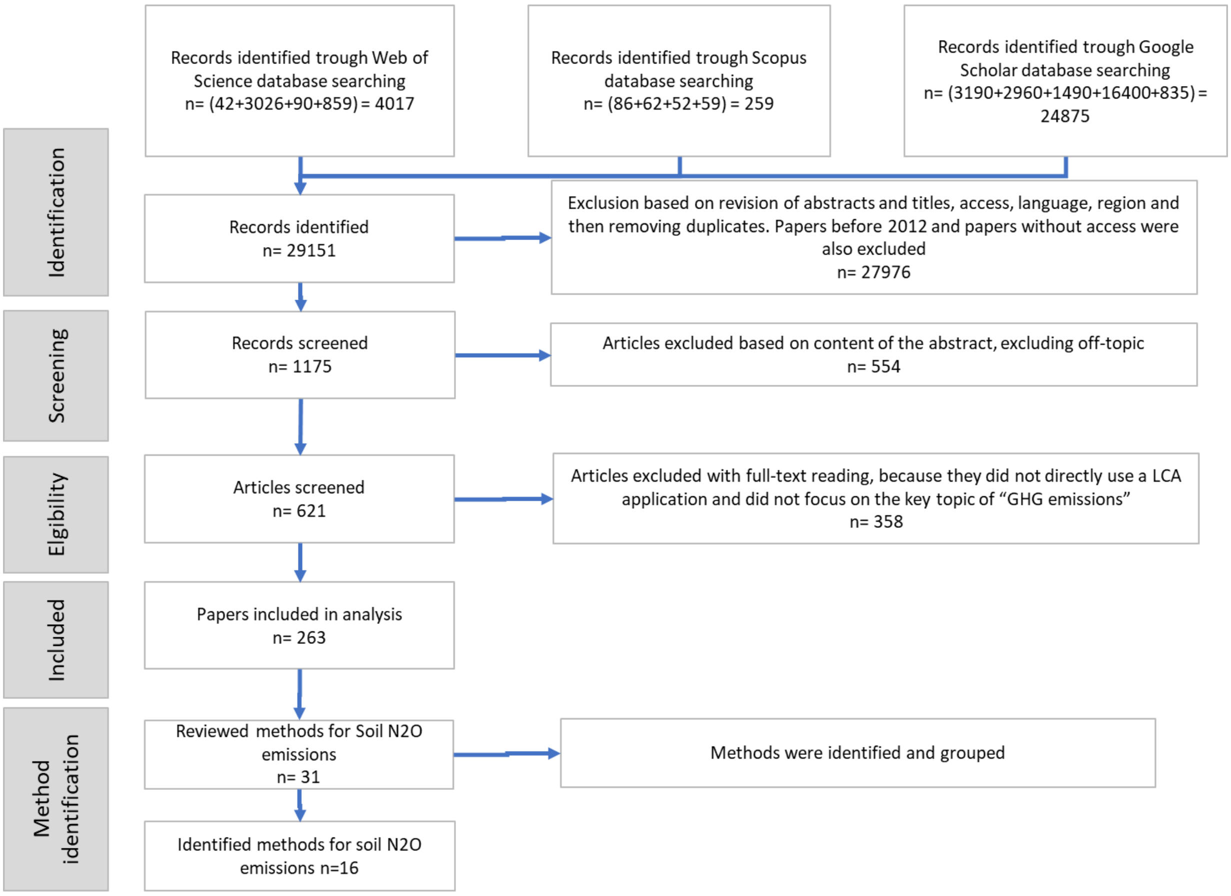


Figure 1: Methodological steps of the literature search process

GENERAL AND SPECIFIC CRITERIA FOR METHOD ASSESSMENT

General criteria

The papers included in this review were then reviewed using both general and specific criteria to assess the LCA methods for crop-livestock systems and products. General criteria used in the harmonization of LCA methods for crop-livestock systems for GHG emissions were selected using a participatory approach based on a modified DELPHI method, extensively described by Goglio et al., (2023a). Briefly, the selection of key topics was carried out through an anonymous survey which allowed us to screen the various topics and provide a priority list on the basis of a preliminary literature review.

General criteria to assess for the LCA methods across the key identified topics were identified. This process began with a review of frameworks used to assess LCA methods. It was undertaken together with articles and publications from literature including the Food and Agriculture Organisation (FAO) Livestock Environmnetal Assessment Programme (LEAP) reports and the Product Environmental

Footprint Category Rules (PEFCR) general guidelines (FAO, 2018; Zampori and Pant, 2019), considering only publicly available sources. Next, an anonymous survey of LCA experts was carried out using Google survey. The general criteria selected through the survey were then further partially reformulated to ensure better consistency and coherence across the key topics selected. Goglio et al., (2023b) describes the general criteria defined for the harmonization of LCA methods for agricultural systems.

Specific criteria identification

Following the definition of the general criteria, specific evaluation criteria were defined for each specific topic in several workshops (n=4). In this paper, only the identification of criteria for soil C and soil N_2O emissions were extensively described. Soil C criteria were here presented as soil C and soil N_2O emissions are closely related (Olesen et al., 2023; Saggar, 2010). However, LCA methods related to soil C in agricultural systems are going to be part of a separate paper. Further information on the specific criteria can be found in Goglio et al. (2023a).
The specific criteria selected for *"Soil C dynamics & Soil N_2O emissions"* are reported in Goglio et al., (2023b) together with their scale: adaptability to different soil types, adaptability for different land uses, and adaptability to different climates. With adaptability to different soil types, we defined the degree at which a LCA method can be applied to different soil types, e.g. peat soils, sandy mineral soils and other type of mineral soils. Instead, with adaptability to land uses, we define the level at which the LCA method can be applied to different land uses (e.g., grassland, cropland); while with the adaptability to different climates, we define the level at which a LCA method can be used in different climatic conditions (e.g., Temperate, Continental, Boreal). Finally, the accuracy was defined as the ability of the LCA methods to capture daily changes and the long-term dynamics of the soil N_2O and CO_2 emissions. With regards to this accuracy definition, it is assumed that the LCA practitioner has sufficient expertise to adopt the methodology and that observations have been carried out with a protocol.

Results

QUANTITATIVE RESULTS

Throughout the systematic review, only 31 LCA methods which assessed soil N_2O emissions in relation to agricultural systems (0.1%) were included in the final review. These LCA methods satisfied most of the general criteria adopted in this research (Figure 2): average score >2.4, across transparency and reproducibility, completeness, fairness and acceptance, robustness criteria (with a scale of 1-4). For these criteria, more than 94% of the LCA methods scored 2 or higher. In contrast, the LCA methods assessed here resulted in low applicability (on average 1.7) with 78% of the LCA methods reviewed in this study scoring 2 or lower with a scale ranging from 1-4 (Figure 2). Four methods scored 3 for applicability: Brentrup et al., 2000, IPCC Tier 1 2006, IPCC Tier 1 2019 methodology and Sozanska et al., (2002) (see section 3.2. for details) (Brentrup et al., 2000; De Klein et al., 2006; Hergoualc'h et al., 2019; Sozanska et al., 2002).

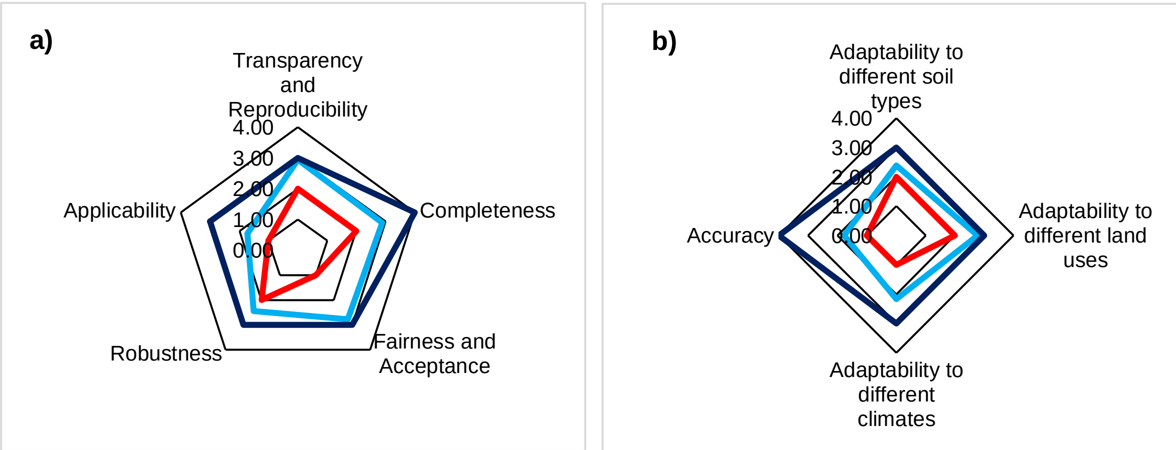


Figure 2: Results from the scoring of the five generic criteria (a) and four specific criteria (b) for LCA methods used to assess soil N2O emissions. Dark blue colour indicates the maximum value obtained, red colour the minimum value and light blue colour the average

Two of the specific criteria were satisfactorily fulfilled (>2.4 on average with a 1-3 scale): adaptability to soil types and land uses. For this set of criteria, all the methods achieved a score of 2 or higher. However, on average, the LCA methods reviewed scored poorly for adapting to different climates and had reasonably low accuracy (<2.2 with a 1-3 scale) (Figure 3). Only four methods scored 3 for adaptability to different climates (IPCC Tier 1 2006 and IPCC Tier 1 2019, DNDC and direct measurements, for details see section 3.2) (De Klein et al., 2006; Li et al., 1996); while 78% scored 2 or less with a range of 1-3. For accuracy, as defined in Goglio et al., (2023b), only direct measurements scored 4, DNDC and DAYCENT scored 3; while most of the methods (83%) scored 2 or lower with a 1-3 scale (Figure 2).

DESCRIPTION AND SCORING OF KEY IDENTIFIED METHODOLOGIES

In this section, a brief description of each identified LCA methodology is presented. The different methods are discussed following a tiered approach as proposed by the IPCC. Three tiers have been proposed following the FAO LEAP framework (FAO, 2020): Simple empirical models and emission factors (Tier 1); Basic process or complex empirical models (Tier 2); Complex process-based models and direct measurements (Tier 3)(FAO, 2020). Direct observations generally fall under the scope of Tier 3 methods, while simple emission factors specific to large geographical areas are Tier 1. The scoring of each method is presented in Table 2.

Simple empirical models and emission factors (Tier 1)

Brentrup This method relies on Bouwman 1995 (Bouwman, 1995), and simply multiplies the total N applied by 1.25% to estimate N_2O emissions for both mineral and organic sources, without distinguishing the source type (Brentrup et al., 2000). This is in contrast with the current IPCC guidelines which uses other values for soil N_2O emissions from both mineral and organic sources (Hergoualc'h et al., 2019). The Brentrup method scored on average 2.6 with the lowest value for the accuracy criterion (1, Table 2).

EMEP/EEA - This method (Amon et al., 2019) was primarily developed for use by national inventory compilers. The authors state that due to its empirical nature, and lack of consideration for site specific soil conditions its use in modelling situations may not be appropriate. It estimates emissions due to manure application and grazing distinguishing between manure types from different livestock categories (Amon et al., 2019). The method scored 2.2 on average among the criteria with lowest values for accuracy, similar to the Brentrup method (1, Table 2).

GLEAM - The updated guidelines (FAO, 2022) for version 3.0 of the GLEAM model provide further guidance on the GLEAM model structure. The equations for N_2O are the same or adapted from the IPCC 2006 or 2019 equations. Thus, they provide a differentiation between crops (based on N biomass content and biological N fixation factors), fertilizer type (ie. manure vs synthetic fertilizer) soil and climatic factors affecting soil N_2O emissions (FAO, 2022). Manure application factors vary slightly from IPCC. Further indirect emissions from leaching losses are estimated based on a nitrogen balance method, different from the IPCC methodology (De Klein et al., 2006; FAO, 2022; Hergoualc'h et al., 2019). GLEAM scored 2.2 and again had the lowest score for the accuracy (1, Table 2).

IPCC (2006) Tier 1 - The Tier 1 method utilizes simple emission factors to estimate direct and indirect N2O emissions, with little differentiation between sources of N (including crop types) or climatic factors (De Klein et al., 2006). With regards to fertilizer management, it distinguishes between mineral and organic fertilizer. IPCC Tier 1 (2006) method scored 2.7 with the lowest value for the accuracy criterion (1, Table 2). IPCC Tier 1 (2019) adopted a broader differentiation between N sources and climatic factors than the IPCC Tier 1 (2006) (De Klein et al., 2006; Hergoualc'h et al., 2019). As shown in Table 2, both IPCC Tier 1 (2019) and IPCC Tier 1 (2006) had the same scores for all the assessed criteria with the smallest value for accuracy (1).

IPCC (2006, 2019) Tier 2 - The Tier 2 method goes beyond Tier 1 through additional differentiation of emissions from synthetic nitrogen types, climatic conditions, and can include country or regional specific Emissions Factors (EFs) (De Klein et al., 2006; Hergoualc'h et al., 2019; Liang et al., 2020).

IPCC Tier 2 had an average score of 2.3, considering both versions together (2006, 2019) and low accuracy values (1, Table 2) (De Klein et al., 2006; Hergoualc'h et al., 2019). Among the methods using the IPCC Tier 2 framework, there is also the most recent version of the Swiss Agricultural LCA (SALCA) method (Nemecek et al., 2023). Regional framework utilizing emission factors were also proposed by Cayuela et al., (2017) for Mediterranean conditions.

Basic process or complex empirical models (Tier 2)

Bonesmo – The Bonesmo et al. 2012 paper utilised IPCC 2006 as the basis for N_2O estimation (Bonesmo et al., 2013), but further refined this into a seasonal (quarterly) estimation, utilising Sozanska et al. (2002)'s data; distinguishing only pasture manure from all other soil N inputs (e.g. organic, synthetic fertilizer, residues). This improved the estimation as it took into account the effects of soil water and temperature on direct N_2O emissions. Indirect N_2O emissions due to nitrate leaching were estimated by multiplying 30% by the total N inputs in kg ha^{-1} and the emission factor derived from IPCC 2006 (De Klein et al., 2006). The Bonesmo method averaged a 2.2 score with the largest value for Transparency and Adaptability to different land uses (3, Table 2).

Holos – Holos is a Canadian model (Little et al., 2008) which has been further enhanced beyond its original scope to use country and regional specific EFs based upon Rochette et al., (2018) and Liang et al., (2020). The model allows for differentiation across crops, soil types, texture and climate by using regional annual precipitation to potential evapotranspiration ratios, allowing for improved emission estimation for Canada. With regards to fertilizer management, it distinguishes between organic and mineral fertilizer (Liang et al., 2020; Rochette et al., 2018). The model had on average a 2.2 score with the lowest applicability value (1, Table 2). Thus, the application of the current Holos version is limited outside Canadian conditions.

INDIGO-N - The Indigo-N v3 model (Bockstaller et al., 2022) provided a new semi-mechanistic approach to estimate nitrate leaching as a source of indirect N_2O losses. While this novel approach tried to account for other losses and agronomic interventions, the direct N_2O emissions, ammonia volatilisation and EFs continued to rely on IPCC Tier 1 or 2 data or equations. These distinguish between different crops (e.g. legumes vs cereals) and fertilizer type (e.g. mineral vs organic). INDIGO N scored as Holos (2.2) with lowest value for applicability (1, Table 2).

INITIATOR - The model INITIATOR (Integrated NITrogen terrestrial systems was partitioned to surface water Impact Assessment Tool On a Regional scale) estimates N_2O emissions through a series of empirical equations estimating nitrification and denitrification (de Vries et al., 2003), and was utilised to assess the impacts of Dutch agriculture on N_2O emissions. INITIATOR distinguish between organic, mineral fertilizer and biological N fixation (de Vries et al., 2003). INITIATOR averaged 2.2 across the assessment criteria and the lowest values were obtained for applicability and adaptability to different climates (1, Table 2).

Sozanska et al. (2002) - This method uses a single regression equation developed for soils in the UK (Sozanska et al., 2002). Whilst this equation was useful as a tool for use with Geographical Information System (GIS) data, the authors acknowledge the uncertainty is high due to the application of short term measured data to arrive at annual estimates. Further, Sozanska et al. (2002)'s method pools all the type of N input together in the calculation. Sozanska et al. (2002)'s method resulted in average score of 2.1 with lowest scores for Fairness and Acceptance (1, Table 2).

Complex process-based models and direct measurements (Tier 3).

CANDY - Utilising the CANDY (CArbon Nitrogen DYnamics) model, a daily time step processing of agricultural soils in 10 cm increments down to 2 m is used for estimating C and N dynamics. These were linked to water and crop sub-models. Two nitrogen forms (nitrate and ammonium) are considered, and processed as nitrogen inputs, conversions and losses, through e.g. nitrification of ammonium to nitrate and denitrification leading to gaseous losses. These are estimated through equations, and related to the crop model for uptake, soil temperature and water (Franko et al., 1995). CANDY averaged 2.4 across the assessment criteria with the highest score for completeness (4, Table 2) and the lowest for accuracy (1, Table 2).

CERES-EGC - The model comprises components to simulate the cycles of water, carbon and nitrogen in agro-ecosystems (Lehuger et al., 2009), and was itself adapted from the semi-empirical NOE model (Goglio et al., 2013). Operating on a daily time step, the agro-ecological model simulates crop development and soil interactions with nitrogen, carbon and water. The N_2O emissions are estimated from 15 parameters, 4 of which require site-specific measurements and the remaining 11 are derived from literature reviews. CERES-EGC had a mean score of 2.6 and had the lowest score for applicability (1, Table 2).

DAYCENT – DayCent was developed as a daily time step version of the CENTURY model (Parton et al., 1994). Daycent is a full agroecological system model, simulating fluxes of C and N between the atmosphere, soil and plant system using a series of empirical equations (Del Grosso et al., 2005; Rotz, 2018). In contrast to e.g. IPCC assumptions of all emissions occurring within the same year of application, DAYCENT includes carry over effects of nitrogen between years and crops. Due to the complex nature of the model, input data goes far beyond simple empirically based calculations, as per other Tier 3 models. DAYCENT had an 2.7 average score and the lowest value for applicability (1, Table 2).

DNDC – DNDC is a mechanistic agroecosystem model (Li et al. 1994), which has been widely used to examine the potential impacts of agricultural management, climate and soils on N_2O emissions, crop yields and other N and C gases. The model includes detailed processes for estimating decomposition, nitrification, denitrification, urea hydrolysis, fermentation and methanogenesis. The model has been shown to perform well in comparison to specific field trials but requires extensive parameterisation to operate under varying soil or climatic conditions (Ehrhardt et al., 2018). DNDC had 2.8 average score and a low applicability value (1, Table 2). Similar to many complex process-based models, DNDC requires extensive expertise and a comprehensive user manual (Gilhespy et al., 2014; Goglio et al., 2018).

ECOSYS - The ECOSYS model allows ecosystem behaviour to be represented in a fully integrated manner under user-defined conditions of soil, climate and management (Welegedara et al., 2020a, 2020b). Of particular relevance to soil N_2O, the soil organic matter microbial populations are represented through five complexes to characterize soil dynamics under varying conditions at an hourly timestep (Metivier et al., 2009). ECOSYS resulted in an average value of 2.4 and the lowest value for applicability (1, Table 2).

Direct measurements Direct measurements have been employed at this stage only for LCA of cropping systems (Goglio et al., 2018), as they are a challenge to be carried out (Laville et al., 2011; Olesen et al., 2023). Direct measurements methods for soil N_2O emissions include chamber, eddy covariance and flux gradient measurements (Glenn et al., 2012; Pattey et al., 2007; Rochette and Eriksen-Hamel, 2008). These methods averaged 2.9 scores despite a low applicability (1) (Table 2).

Group	LCA publication[a]	Method name	Method publication[b]	General criteria*					Specific criteria				
				Trans. and Rep.	Com.	Fair. and Accept.	Robust.	App.	Adapt. to different soil types	Adapt. to different land uses	Adapt. to different climates	Acc.	Mean Score
Simple empirical models and emission factors (Tier 1)	Schmidt Rivera et al., (2017)	Brentrup 2000	Brentrup et al., (2000)	3	3	3	3	3	2	3	2	1	2.56
	Berton et al., (2016)	EEA 2013 (2019 reviewed)	Amon et al., (2019)	3	3	3	2	2	2	2	2	1	2.22
	MacLeod et al., (2018)	GLEAM model. IPCC 2006 tier 2 combined with LCA analysis	FAO, (2017)	3	3	3	2	2	2	2	2	1	2.22
	Cederberg et al., (2013)	IPCC (2006) Tier 1	Lasco et al., (2006)	3	3	3	2	3	3	3	3	1	2.67
	Jeswani et al., (2018)	IPCC (2006) Tier 2	De Klein et al., (2006)	3	3	3	2	2	2	3	2	1	2.33
	González-Quintero et al., (2021)	IPCC 2006 (2019 refinement) Tier 1	Hergoualc'h et al., (2019)	3	3	3	2	3	3	3	3	1	2.67
Basic process or complex empirical models (Tier 2)	Bonesmo et al., (2013)	Bonesmo et al 2012	Bonesmo et al., (2012)	3	2	2	2	2	2	3	2	2	2.22
	Alemu et al., (2017)	Holos	Little et al., (2008)	3	3	3	2	1	2	2	2	2	2.22
	Avadí, (2020)	INDIGO-N combined with IPCC Tier 1 emission factors	Bockstaller et al., (2022)	3	3	3	2	1	2	2	2	2	2.22
	de Vries et al., (2015)	INITIATOR	de Vries et al., (2003)	3	2	2	3	1	3	3	1	2	2.22
	Bonesmo et al., (2013)	Sozanska et al.(2002)	Sozanska et al., (2002)	2	2	1	2	3	2	3	2	2	2.11
Complex process based models models and direct	Carauta et al., (2021)	CANDY	Franko et al., (1995)	3	4	3	2	2	2	3	2	1	2.44
	Cederberg et al., (2013)	CERES-EGC	Goglio et al., (2013)	3	3	3	3	1	3	3	2	2	2.56
	Cederberg et al., (2013)	DAYCENT	Del Grosso et al., (2005)	3	3	3	3	1	3	3	2	2	2.56
	Grossi et al., (2021)	DNDC	Li et al., (1994)	3	3	3	3	1	3	3	3	3	2.78

Rotz, (2018)	ECOSYS	Metivier et al., (2009)	3	3	3	3	1	2	3	2	2	2.44
[c]	Direct observations	[c, d]	3	3	3	3	1	3	3	3	4	2.89

Table 2: Details of the described methods, including the general criteria and specific criteria scoring.

**General criteria abbreviations: Trans.: Transparency; Rep.: Reproducibility; Com.: Completeness; Fair.: Fairness; Accept.: Acceptance; Robust.: Robustness; App.: Applicability; Adapt. : Adaptability; Acc.: Accuracy*

[a]Publications where the method has been used in the LCA of agricultural systems

[b]Key publication where the method has been extensively described

[c]no publications used direct observation in LCA of livestock systems, however a LCA of cropping systems used direct observations (Goglio et al., 2018c).

[d]Several research studies discussed direct N_2O observations techniques (Glenn et al., 2012; Pattey et al., 2007; Rochette et al., 2018; Rochette and Eriksen-Hamel, 2008; Venterea et al., 2020).

ASSESSMENT OF THE LCA METHODS FOR SOIL N_2O EMISSIONS

The assessment of the LCA methods was carried out by providing a scoring for the general and specific criteria. The results of the assessment were discussed among the group of experts (n=14) in a series of workshops (n=22), then they were further reviewed by other experts external to the PATHWAYS project. The identified experts had expertise in LCA of agricultural systems and soil N_2O emission quantification. All the discussions were conducted as a community of peers among experts (Macombe et al., 2018), in line with the harmonisation approach for LCA of livestock systems and products (Goglio et al., 2023a). Targeted and structured discussions were organised to solve eventual disagreement in the scoring of the LCA methods, as previously carried out (Goglio et al., 2023a; Macdiarmid et al., 2016).

Discussion

IDENTIFIED KEY METHODOLOGICAL ISSUES

The LCA methods assessed in the present review of soil N_2O emissions were transparent and easy to reproduce, complete, robust, fair and accepted. However, a large proportion have low applicability (50%) and accuracy (39%), whilst the majority of the methods (78%) had low adaptability to different climates. The five methods with very high applicability (3) were Brentrup et al. (2000); Sozanska et al., (2002); IPCC Tier 1 methodology 2006 and 2019 (De Klein et al., 2006; Hergoualc'h et al., 2019) and direct measurements. Brentrup et al., (2000), IPCC Tier 1 (2006) and IPCC Tier 1 (2019) were probably the more general methods which could be applied for every condition, soil climate, soil type and soil management, though they do not include the effects of nitrification inhibitors, slow release fertilizer, timing of fertilizer or manure applications, and type of spreading and distribution for manure and slurry (De Klein et al., 2006; Hergoualc'h et al., 2019). On the other hand, Sozanska et al., 2002's method was generally more accurate, however it required very specific data such as water filled spore space (WFPS) measured directly from the field (Bastos et al., 2021; Rochette et al., 2018; Venterea et al., 2011), which is often not available to the LCA practitioner. However, together with WFPS, during the season, different conditions have to be verified for the soil N_2O emissions to occur such as high nitrate availability and high temperature. While assessing accuracy, these aspects were taken into account as these soil parameters are subject to daily changes which affect emissions (Bastos et al., 2021; Saggar, 2010).

Group	LCA publication[a]	Method name	Method publication[b]	General criteria: Trans. and Rep.	Com.	Fair. and Accept.	Robust.	App.	Specific criteria: Adapt. to different soil types	Adapt. to different land uses	Adapt. to different climates	Acc.	Mean Score
Simple empirical models and emission factors (Tier 1)	Schmidt Rivera et al., (2017)	Brentrup 2000	Brentrup et al., (2000)	3	3	3	3	3	2	3	2	1	2.6
	Berton et al., (2016)	EEA 2013 (2019 reviewed)	Amon et al., (2019)	3	3	3	2	2	2	2	2	1	2.2
	MacLeod et al., (2018)	GLEAM model. IPCC 2006 tier 2 combined with LCA analysis	FAO, (2017)	3	3	3	2	2	2	2	2	1	2.2
	Cederberg et al., (2013)	IPCC (2006) Tier 1	Lasco et al., (2006)	3	3	3	2	3	3	3	3	1	2.7
	Jeswani et al., (2018)	IPCC (2006) Tier 2	De Klein et al., (2006)	3	3	3	2	2	2	3	2	1	2.3
	González-Quintero et al., (2021)	IPCC 2006 (2019 refinement) Tier 1	Hergoualc'h et al., (2019)	3	3	3	2	3	3	3	3	1	2.7
Basic process or complex empirical models (Tier 2)	Bonesmo et al., (2013)	Bonesmo et al 2012	Bonesmo et al., (2012)	3	2	2	2	2	2	3	2	2	2.2
	Alemu et al., (2017)	Holos	Little et al., (2008)	3	3	3	2	1	2	2	2	2	2.2
	Avadí, (2020)	INDIGO-N combined with IPCC Tier 1 emission factors	Bockstaller et al., (2022)	3	3	3	2	1	2	2	2	2	2.2
	de Vries et al., (2015)	INITIATOR	de Vries et al., (2003)	3	2	2	3	1	3	3	1	2	2.2
	Bonesmo et al., (2013)	Sozanska et al.(2002)	Sozanska et al., (2002)	2	2	1	2	3	2	3	2	2	2.1
Complex process based models and direct models	Carauta et al., (2021)	CANDY	Franko et al., (1995)	3	4	3	2	2	2	3	2	1	2.4
	Cederberg et al., (2013)	CERES-EGC	Goglio et al., (2013)	3	3	3	3	1	3	3	2	2	2.6
	Cederberg et al., (2013)	DAYCENT	Del Grosso et al., (2005)	3	3	3	3	1	3	3	2	3	2.7
	Grossi et al., (2021)	DNDC	Li et al., (1994)	3	3	3	3	1	3	3	3	3	2.8

Rotz, (2018)	ECOSYS	Metivier et al., (2009)	3	3	3	3	1	2	3	2	2	2.4
[c]	Direct observations	[c, d]	3	3	3	3	1	3	3	3	4	2.9

Table 2 Details of the described methods, including the general criteria and specific criteria scoring. Trans.: Transparency; Rep.: Reproducibility; Com.: Completeness; Fair.: Fairness; Accept.: Acceptance; Robust.: Robustness; App.: Applicability; Adapt. : Adaptability; Acc.: Accuracy

[a]Publications where the method has been used in the LCA of agricultural systems

[b]Key publication where the method has been extensively described

[c]The LCA method assessment was based on a LCA of cropping systems using direct observations (Goglio et al., 2018).

[d]Several research studies discussed direct N_2O observations techniques (Glenn et al., 2012; Pattey et al., 2007; Rochette et al., 2018; Rochette and Eriksen-Hamel, 2008; Venterea et al., 2020).

The LCA based on DNDC, DAYCENT or direct measurements scored 3 in accuracy, as DNDC and DAYCENT accounts for soil moisture and temperature, soil C and N dynamics and crop N uptake at a daily time step (Brilli et al., 2017; Del Grosso et al., 2020; Ehrhardt et al., 2018; Giltrap et al., 2020; Li et al., 1996). Instead, while the direct measurements performed very well for all the criteria except applicability, their use in a LCA is limited by the difficulties in carrying out the monitoring both from a technical and financial stand point (Dorich et al., 2020; Giltrap et al., 2020; Laville et al., 2011; Olesen et al., 2023), which make these data hardly available to the LCA practitioner. Further, it may be challenging to allocate the soil N_2O emissions related to a particular crop management since the impacts can carry over to the period when the following crop is grown as discussed previously for crop residues (Goglio et al., 2017; Olesen et al., 2023). As for LCA method for soil C in agricultural LCA (Goglio et al., 2015), a compromise also has to be found between accuracy and applicability of the LCA method for soil N_2O. This compromise is dependent on data availability, LCA practitioner expertise in coherence with the LCA objectives (Goglio et al., 2015). In some cases, simpler methods for estimating N_2O emissions may not include some field management practices (e.g. impacts of urease and nitrification inhibitors, split fertilizer application, or N credit from legumes).

Most of the methods assessed fit into two categories: IPCC Tier 1 methodology and subsequent updates or agroecosystem models, such as DNDC and DAYCENT (Del Grosso et al., 2005; Gilhespy et al., 2014; Goglio et al., 2018). Different from soil C, empirical or regression models are currently not available, except those proposed by Sozanska et al., (2002), however this latter method was developed for the Atlantic climate and depends on data which were rarely available to the common LCA practitioner. Other methods were developed to estimate soil N_2O emissions based on parameters such as rainfall, soil characteristics and N management in Canadian conditions (Rochette et al., 2018), which could be used in the LCA of livestock systems. This type of data is more commonly used and collected in agricultural LCA (Goglio et al., 2018; Styles et al., 2014).

For large scale site-dependent assessment, either attributional, consequential or anticipatory LCA, using the IPCC Tier 1 methodology (using the 2019 updated and disaggregated by climate type values) is a sensible compromise between the accuracy and the applicability of the LCA method. For countries where IPCC Tier 2 emission factors are available, the latter methodology should be preferred as it is more accurate in capturing local conditions (Cayuela et al., 2017; Hergoualc'h et al., 2019), however it might be challenging to collect data with enough quality to use IPCC emission factors in both cropping, grassland, agricultural and livestock systems. Indeed for the latter, a higher level of system complexity is achieved as feed (e.g. cropping systems) and fodder (e.g. grassland systems) producing systems need to be assessed (Rotz, 2018).

Improving soil N_2O emissions quantification is important as N2O impacts global warming, but can also contribute and affect other impact categories in combination with other important emissions such as ammonia and NOx. These impact categories can include biodiversity loss, stratospheric ozone-depletion, eutrophication (which is related to water quality degradation) and acidification (which is affected by air pollution). Indeed, the effects on climate change (i.e. global warming) can alter indirectly several of the ecosystem services provided by the cropping and grassland systems, including water availability (Brady and Weil, 2002; Hergoualc'h et al., 2019; Pörtner et al., 2022).

Research need, future studies

Soil N_2O emissions derived from both soil tillage management and fertilizer management including manure, sludge or slurry spreading are often dependent on the interaction between soil characteristics, rainfall and temperature (Bastos et al, 2021; Saggar 2010). While the pattern of soil N_2O emissions related to the mineral and organic fertilizer application is rather well known (Dorich et al., 2020; Giltrap et al., 2020; Taki et al., 2019), the interaction with residues from legume crops is less clear (Chirinda et al., 2010; Olesen et al., 2023). The latter together with grassland and cover crop management are particularly important in livestock systems (Parajuli et al., 2018).

Within the LCA context, there is a general need to ensure that crop and grassland management issues are considered and accurately accounted for in the LCA of agricultural systems, as previously discussed for organic agriculture (van der Werf et al., 2020). This is in view of pollution shifts and trade-off across impact categories, related to the N biogeochemical cycle (Brady and Weil, 2002; Styles et al., 2015; Zhou et al., 2023). With regards to soil N_2O emissions, only the DNDC was able to fully capture soil N_2O drivers, crop management, soil and climate characteristics (Brilli et al., 2017; Del Grosso et al., 2020; Li et al., 1996). However its applicability is low due to a large data requirement and a need for modeller expertise, as previously discussed for soil C (Giltrap et al., 2020).

Emissions from crop residues can happen during the growing season of the following crop, when high biomass is degraded and high water content is available (Olesen et al., 2023). This can cause allocation issues among crops in agricultural LCA, thus a system approach might be necessary, as previously discussed (Goglio et al., 2017; Sieverding et al., 2020). However, even with a system approach in agricultural LCA, the environmental impacts from a specific crop within a specific cropping system should be allocated, if the latter is used as feed in a livestock system (Rotz, 2018).

Therefore, soil N_2O emission methods need to be developed to capture crop management effects on soil N_2O emissions, similar to DNDC, without limitations from data requirements. An option is the method by Sozanska et al. (2002), even though it did not capture many aspects of crop management such as tillage, residue management, type of fertilizer, rainfall patterns (Bastos et al., 2021; Saggar, 2010). Alternative methods could be based on statistical methods used for gap-filling (e.g., random forest or neural networks), which use a series of covariates factors to estimate soil N_2O emissions (Dorich et al., 2020). On the other hand, regression models, similar to those developed in Rochette et al., (2018), which capture more aspects of crop and grassland management, such as crop type, some fertilizer management, soil and climate characteristics, should be developed for European conditions.

These could be a compromise between accuracy of the model and applicability of the methods for LCA of agricultural systems.
Further, soil N2O emission play an important contribution to the overall global GHG budget (0.75%) (IPCC, 2022). Thus, efforts should be made to improve the estimates by increasing the available data across Europe and by comparing agroecosystem model performance of the key models identified here (ie. DNDC and DAYCENT) to better improve the overall GHG estimates. Previously a metanalysis was carried out in Canadian conditions (Liang et al., 2020) and a similar analysis could be performed in Europe by assessing all scientific evidence related to the impact on soil N_2O emissions due to crop/grassland management practices, soil and climate conditions (Liang et al., 2020; Rochette et al., 2018). This research would contribute to the overall improvement of the IPCC GHG emission calculations (Hergoualc'h et al., 2019).

LCA recommendations

Our review of soil N_2O emissions leads to the recommendation that it is preferable to use the DNDC model after calibration and validation or use of direct field measurements, taking in consideration system effects (Goglio et al., 2017). However, when the necessary data to run the DNDC model or field observations are lacking, the use of IPCC tier 2 methodology (2019) with disaggregated EFs should be prioritized where available, otherwise IPCC Tier 1 methodology following the 2019 guidelines should be used (Hergoualc'h et al., 2019). When using 2019 IPCC Tier 2 or IPCC Tier 1 methodology to assess soil N_2O emissions, the methodological limitations should be made clear by the LCA practitioner (Hergoualc'h et al., 2019). Independently from the methodological choice carried out, it is key to provide arguments for this choice and describe its potential limitations, in agreement with the ISO standards (ISO, 2006a, 2006b, 2013).
Especially for large site-dependent or site-generic studies (Potting and Hauschild, 2006), a preliminary assessment could still be carried out using simpler methods such as IPCC Tier 1 (2019) (Hergoualc'h et al., 2019), as data might not be available for the LCA practitioner. This should be complemented with a clear description of limitations of the methodology as suggested by the ISO standards (ISO, 2006b, 2006a, 2013) and discussed in the present research. Further, conclusions about these LCAs should be taken with caution as they poorly reflect local conditions and the effect of crop and grassland management. Indeed, local conditions are key in soil N_2O emissions as these are subject to a large spatial variability (Del Grosso et al., 2020).
This harmonization of LCA methods has been carried out with a participatory approach involving several experts (n=14) which have been involved at different stage of the process, following the criteria previously drawn (Goglio et al., 2023a). This approach allowed for the development of scoring criteria to assess LCA methods through workshops and targeted discussion, as previously discussed for social LCA (Macombe et al., 2018). This harmonization approach allowed for the discussion of state

of the art practices and the identification of future development priorities and future needs in a coherent manner for several topics including soil C, manure emissions, enteric fermentation, biodiversity, animal welfare, nutrition aspects and circular economy (Goglio et al., 2023a).

Conclusion

In this research an attempt to harmonize LCA methods for soil N_2O emissions in agricultural systems was carried out by comparing methods, showing their limitations and making recommendation on their use. It was observed that a high level of accuracy corresponded to a low level of applicability and vice versa. Thus, the choice of the methodology in relation to the LCA objectives is particularly critical to enable high quality LCA assessments.

Following the analysis of the available literature, series of recommendations was proposed. A general recommendation for soil N_2O from agricultural systems is that the choice of LCA methods should be based on the LCA objectives, data availability and expertise of the LCA practitioner. For all soil N_2O assessments, more complex methods are available but have greater data requirements. IPCC Tier 1 methodology has been employed in most of the assessments analysed here. Independently of the method used, method limitations should be discussed in the LCA of agricultural systems in view of the assessment objectives, data requirements and expertise available. Further, within the IPCC, there is a urgent need to develop higher Tier methods to improve the overall assessment of soil N_2O emissions. This could be achieved to a broader testing and comparison of field observations with the models identified here to improve the IPCC methodology. This research should be combined with a metanalysis of all the drivers affecting soil N2O emissions in cropland/grassland systems.

Future development of LCA methodology is necessary to improve LCA of agricultural systems. For soil N_2O emission, effort should be placed towards developing a basic process model (i.e. soil N_2O regression models) which optimises applicability and accuracy. The LCA method development related to soil N_2O emissions must be synchronous with improvements of quantification methods and the assessment of different agricultural management.

Funding sources

This research has been developed within the PATHWAYS project, funded by the European Union's Horizon 2020 Research and Innovation Programme under grant agreement No 101000395.

References

Alemu, A.W., Amiro, B.D., Bittman, S., MacDonald, D., Ominski, K.H., 2017. Greenhouse gas emission of Canadian cow-calf operations: A whole-farm assessment of 295 farms. Agr. Syst. 151, 73–83 doi: 10.1016/j.agsy.2016.11.013.

Amon, B., Hutchings, N., Dämmgen, U., Sven Sommer, J.W., Seedorf, J., Hinz, T., Hoek, K., Gyldenkærne, S., Mikkelsen, M.H., Dore, C., Jiménez, B.S., Menzi, H., Dedina, M., Haenel, H.-D., Röseman, C., Groenestein, K., Bittman, S., Hobbs, P., Lekkerkerk, L., Bonazzi, G., Couling, S., Cowell, D., Kroeze, C., Pain, B., Klimont, Z., 2019.3.B Manure management. (EMEP/EEA air pollutant emission inventory Guidebook 2019). EEA European Environmental Agency, Copenhagen, Denmark.

Avadí, A., 2020. Screening LCA of French organic amendments and fertilisers. Int. J. Life Cycle Assess. 25, 698–718 doi: 10.1007/s11367-020-01732-w.

Bastos, L.M., Rice, C.W., Tomlinson, P.J., Mengel, D., 2021. Untangling soil-weather drivers of daily N2O emissions and fertilizer management mitigation strategies in no-till corn. Soil Sci. Soc. Am. J. 85, 1437–1447 doi: 10.1002/saj2.20292.

Berton, M., Cesaro, G., Gallo, L., Pirlo, G., Ramanzin, M., Tagliapietra, F., Sturaro, E., 2016. Environmental impact of a cereal-based intensive beef fattening system according to a partial Life Cycle Assessment approach. Livest. Sci. 190, 81–88 doi: 10.1016/j.livsci.2016.06.007.

Bockstaller, C., Galland, V., Avadí, A., 2022. Modelling direct field nitrogen emissions using a semi-mechanistic leaching model newly implemented in Indigo-N v3. Ecol. Model. 472, 110109 doi: 10.1016/j.ecolmodel.2022.110109.

Bonesmo, H., Skjelvåg, A.O., Henry Janzen, H., Klakegg, O., Tveito, O.E., 2012. Greenhouse gas emission intensities and economic efficiency in crop production: A systems analysis of 95 farms. Agr. Syst. 110, 142–151 doi: 10.1016/j.agsy.2012.04.001.

Bonesmo, H., Beauchemin, K.A., Harstad, O.M., Skjelvåg, A.O., 2013. Greenhouse gas emission intensities of grass silage based dairy and beef production: A systems analysis of Norwegian farms. Livest. Sci. 152, 239–252 doi: 10.1016/j.livsci.2012.12.016.

Bouwman, A.E., 1995.Compilation of a global inventory of emissions of Nitrous Oxide (Ph.D. thesis).

Brady, N., Weil, R., 2002.The Nature and Properties of Soils, 13th ed. Prentice Hall, Upper Saddle River, New Jersey, USA.

Brentrup, F., Küsters, J., Lammel, J., Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. Int. J Life Cycle Assess. 5, 349 doi: 10.1007/BF02978670.

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S., 2013. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philosophical Transactions of the Royal Soc. B: Biol. Sci. 368, 20130122 doi: 10.1098/rstb.2013.0122.

Cabot, M.I., Lado, J., Bautista, I., Ribal, J., Sanjuán, N., 2023. On the relevance of site specificity and temporal variability in agricultural LCA: a case study on mandarin in North Uruguay. Int. J. Life Cycle Assess. 28, 1516–1532 doi: 10.1007/s11367-023-02186-6.

Carauta, M., Troost, C., Guzman-Bustamante, I., Hampf, A., Libera, A., Meurer, K., Bönecke, E., Franko, U., Ribeiro Rodrigues, R. de A., Berger, T., 2021. Climate-related land use policies in Brazil: How much has been achieved with economic incentives in agriculture? Land Use Policy 109, 105618 doi: 10.1016/j.landusepol.2021.105618.

Cayuela, M.L., Aguilera, E., Sanz-Cobena, A., Adams, D.C., Abalos, D., Barton, L., Ryals, R., Silver, W.L., Alfaro, M.A., Pappa, V.A., Smith, P., Garnier, J., Billen, G., Bouwman, L., Bondeau, A., Lassaletta, L., 2017. Direct nitrous oxide emissions in Mediterranean climate cropping systems: Emission factors based on a meta-analysis of available measurement data. Agr. Ecosys. Environ. 238, 25–35 doi: 10.1016/j.agee.2016.10.006.

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. Anim. 7, 212–219 doi: 10.1017/S1751731113000785.

Chirinda, N., Carter, M., Albert, K., Ambus, P., Olesen, J.E., Porter, J.R., Petersen, S.O., 2010. Emissions of nitrous oxide from arable organic and conventional cropping systems on two soil types. Agr. Ecosyst. Environ. 136, 199–208 doi: 10.1016/j.agee.2009.11.012.

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 doi: 10.1007/s11367-020-01812-x.

De Klein, C., Novoa, R.S.A., Ogle, S., Smith, K.A., Rochette, P., Wirth, T.C., McConkey, B.G., Mosier, A., Rypdal, K., Walsh, M., Williams, S.A., 2006. Chapter 11: N2O Emissions from Managed Soils, and CO2 Emissions from Lime and Urea Application, in: Gytarsky M, Hiraishi T, Irving W, Krug T, Penman J Editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC International Panel on Climate Change, Geneva, p. 11.1-11.54.

de Vries, W., Kros, J., Oenema, O., de Klein, J., 2003. Uncertainties in the fate of nitrogen II: A quantitative assessment of the uncertainties in major nitrogen fluxes in the Netherlands. Nutr. Cycl. Agroecosystems 66, 71–102 doi: 10.1023/A:1023354109910.

de Vries, W., Kros, J., Dolman, M.A., Vellinga, Th.V., de Boer, H.C., Gerritsen, A.L., Sonneveld, M.P.W., Bouma, J., 2015. Environmental impacts of innovative dairy farming systems aiming at improved internal nutrient cycling: A multi-scale assessment. Sci. Tot. Environ. 536, 432–442 doi: 10.1016/j.scitotenv.2015.07.079.

Del Grosso, S.J., Mosier, A., Parton, W., Ojima, D., 2005. DAYCENT model analysis of past and contemporary soil NO and net greenhouse gas flux for major crops in the USA. Soil. Till. Res. 83, 9–24 doi: 10.1016/j.still.2005.02.007.

Del Grosso, S.J., Smith, W., Kraus, D., Massad, R.S., Vogeler, I., Fuchs, K., 2020. Approaches and concepts of modelling denitrification: increased process understanding using observational data can reduce uncertainties. Current Opinion in Environ. Sustain. 47, 37–45 doi: 10.1016/j.cosust.2020.07.003.

Dorich, C.D., De Rosa, D., Barton, L., Grace, P., Rowlings, D., Migliorati, M.D.A., Wagner-Riddle, C., Key, C., Wang, D., Fehr, B., Conant, R.T., 2020. Global Research Alliance N2O chamber methodology guidelines: Guidelines for gap-filling missing measurements. J. Environ. Qual. 49, 1186–1202 doi: https://doi.org/10.1002/jeq2.20138.

Ehrhardt, F., Soussana, J.-F., Bellocchi, G., Grace, P., McAuliffe, R., Recous, S., Sándor, R., Smith, P., Snow, V., de Antoni Migliorati, M., Basso, B., Bhatia, A., Brilli, L., Doltra, J., Dorich, C.D., Doro, L., Fitton, N., Giacomini, S.J., Grant, B., Harrison, M.T., Jones, S.K., Kirschbaum, M.U.F., Klumpp, K., Laville, P., Léonard, J., Liebig, M., Lieffering, M., Martin, R., Massad, R.S., Meier, E., Merbold, L., Moore, A.D., Myrgiotis, V., Newton, P., Pattey, E., Rolinski, S., Sharp, J., Smith, W.N., Wu, L., Zhang, Q., 2018. Assessing uncertainties in crop and pasture ensemble model simulations of productivity and N2O emissions. Glob. Change Biol. 24, e603–e616 doi: 10.1111/gcb.13965.

Ershadi, S.Z., Dias, G., Heidari, M.D., Pelletier, N., 2020. Improving nitrogen use efficiency in crop-livestock systems: A review of mitigation technologies and management strategies, and their potential applicability for egg supply chains. J. Clean. Prod. 265, 121671 doi: 10.1016/j.jclepro.2020.121671.

FAO, 2017.Global Livestock Environmental Assessment Model Version 2.0 Model description revision 6. Food and Agriculture Organisation of the United Nations, Rome, Italy.

FAO, 2018.Measuring and modelling soil carbon stocks and stock changes in livestock production systems – Guidelines for assessment (Draft for public review). Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2020.Livestock Environmental Assessment and Performance (LEAP) Partnership. FAO, Food and Agriculture Organization of the United Nations. http://www.fao.org/partnerships/leap/en/ (accessed 11 may 2020).

FAO, 2022.Global Livestock Environmental Assessment Model (GLEAM). https://www.fao.org/gleam/en/ (accessed 9 august 2022).

FAO, 2023.Pathways towards lower emissions A global assessment of the greenhouse gas emissions and mitigation options from livestock agrifood systems. Food and Agriculture Organisation of the United Nations, Rome, Italy.

Franko, U., Oelschlägel, B., Schenk, S., 1995. Simulation of temperature-, water- and nitrogen dynamics using the model CANDY. Ecol. Model. 81, 213–222 doi: 10.1016/0304-3800(94)00172-E.

Gilhespy, S.L., Anthony, S., Cardenas, L., Chadwick, D., Del Prado, A., Li, C., Misselbrook, T., Rees, R.M., Salas, W., Sanz-Cobena, A., Smith, P., Tilston, E.L., Topp, C.F.E., Vetter, S., Yeluripati, J.B.,

2014. First 20 years of DNDC (DeNitrification DeComposition): Model evolution. Ecol. Model. 292, 51–62 doi: 10.1016/j.ecolmodel.2014.09.004.

Giltrap, D., Yeluripati, J., Smith, P., Fitton, N., Smith, W., Grant, B., Dorich, C.D., Deng, J., Topp, C.F., Abdalla, M., Liáng, L.L., Snow, V., 2020. Global Research Alliance N2O chamber methodology guidelines: Summary of modeling approaches. J. Environ. Qual. 49, 1168–1185 doi: 10.1002/jeq2.20119.

Glenn, A.J., Tenuta, M., Amiro, B.D., Maas, S.E., Wagner-Riddle, C., 2012. Nitrous oxide emissions from an annual crop rotation on poorly drained soil on the Canadian Prairies. Agr. Forest Meteorol. 166–167, 41–49 doi: 10.1016/j.agrformet.2012.06.015.

Godfray, H.C.J., Aveyard, P., Garnett, T., Hall, J.W., Key, T.J., Lorimer, J., Pierrehumbert, R.T., Scarborough, P., Springmann, M., Jebb, S.A., 2018. Meat consumption, health, and the environment. Sci. 361, eaam5324 doi: 10.1126/science.aam5324.

Goglio, P., Colnenne-David, C., Laville, P., Doré, T., Gabrielle, B., 2013. 29% N2O emission reduction from a modelled low-greenhouse gas cropping system during 2009–2011. Environ. Chem. Letters 11, 143–149 doi: 10.1007/s10311-012-0389-8.

Goglio, P., Grant, B.B., Smith, W.N., Desjardins, R.L., Worth, D.E., Zentner, R., Malhi, S.S., 2014. Impact of management strategies on the global warming potential at the cropping system level. Sci. Tot. Environ. 490, 921–933 doi: 10.1016/j.scitotenv.2014.05.070.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J. Clean. Prod. 104, 23–39 doi: 10.1016/j.jclepro.2015.05.040.

Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2017. Addressing crop interactions within cropping systems in LCA. Int. J. Life Cycle Assess. 1–9 doi: 10.1007/s11367-017-1393-9.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., Gao, X., Hanis, K., Tenuta, M., Campbell, C.A., McConkey, B.G., Nemecek, T., Burgess, P.J., Williams, A.G., 2018. A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. J. Clean. Prod. 172 doi: 10.1016/j.jclepro.2017.03.133.

Goglio, P., Knudsen Trydeman, M., 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., 2023a. Defining common criteria for harmonizing life cycle assessments of livestock systems. Clean. Prod. Letters 4, 100035 doi: 10.1016/j.clpl.2023.100035.

[dataset] Goglio, P., Moakes, S., Trydeman Knudsen, M., Van Mierlo, V., Röhrig, N., Maxime, F., Maresca, A., Romero-Huelva, M., Waqas, M.A., Laurence G. Smith, Grossi, G., Smith, W., De Camillis, C., Nemecek, T., 2023b. Criteria for LCA assessment of soil N2O emissions in agricultural systems doi: 10.5281/zenodo.10006380.

González-Quintero, R., Bolívar-Vergara, D.M., Chirinda, N., Arango, J., Pantevez, H., Barahona-Rosales, R., Sánchez-Pinzón, M.S., 2021. Environmental impact of primary beef production chain in Colombia: Carbon footprint, non-renewable energy and land use using Life Cycle Assessment. Sci. Tot. Environ. 773, 145573 doi: 10.1016/j.scitotenv.2021.145573.

Grossi, G., Goglio, P., Vitali, A., Williams, A.G., 2019. Livestock and climate change: impact of livestock on climate and mitigation strategies. Anim. Frontiers 9, 69–76 doi: 10.1093/af/vfy034.

Grossi, G., Vitali, A., Bernabucci, U., Lacetera, N., Nardone, A., 2021. Greenhouse Gas Emissions and Carbon Sinks of an Italian Natural Park. Front. Environ. Sci. 9.

Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., Del Prado, N., Kasimir, A., MacDonald, D., Ogle, S., Regina, K., van der Weerden, T., Liang, C., Noble, A., 2019. N2O emissions from managed soils, and CO2 emissions from lime and urea application, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Interngovernmental Panel for Climate Change, Geneva.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., Zelm, R. van, 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J Life Cycle Assess. 22, 138–147 doi: 10.1007/s11367-016-1246-y.

IPCC, 2022. Climate change 2022: Mitigation of climate change. WGIII Mitigation of Climate Change Climate Change 2022 Working Group III contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, Interngovernmental panel for climate change, Geneva, Switzerland.

ISO, 2006a.SS-EN ISO 14040 Environmental Management- Life Cycle Assessment, Principles and Framework. International Organization for Standardization, Geneva.

ISO, 2006b.SS-EN ISO 14044 Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organization for Standardization, Geneva.

ISO, 2013.TS-EN ISO 14067 Greenhouse Gases -Carbon Footprint of Products- Requirements and Guidelines for Quantification and Communication. International Organization for Standardization, Geneva.

Jeswani, H.K., Espinoza-Orias, N., Croker, T., Azapagic, A., 2018. Life cycle greenhouse gas emissions from integrated organic farming: A systems approach considering rotation cycles. Sustain. Prod. Consum. 13, 60–79 doi: 10.1016/j.spc.2017.12.003.

Jourdaine, M., Loubet, P., Trebucq, S., Sonnemann, G., 2020. A detailed quantitative comparison of the life cycle assessment of bottled wines using an original harmonization procedure. J. Clean. Prod. 250, 119472 doi: 10.1016/j.jclepro.2019.119472.

Kimming, M., Sundberg, C., Nordberg, Å, Baky, A., Bernesson, S., Norén, O., Hansson, P.-A., 2011. Biomass from agriculture in small-scale combined heat and power plants – A comparative life cycle assessment. Biomass Bioenerg. 35, 1572–1581 doi: 10.1016/j.biombioe.2010.12.027.

Lasco, R., Ogle, S., Raison, J., Verchot, L., Wassman, R., Yagi, K., Bhattacharya, S., Brenner, J., Partson Daka, J., Gonzalez, S., Krug, T., Li, Y., Martino, D., McConckey, B., Smith, P., Tyler, S., Zhakata, W., Sass, R., Yan, X., 2006. Chapter 5 Cropland, in: Gytarsky M, Hiraishi T, Irving W, Krug T, Penman J Editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC International Panel on Climate Change, Geneva.

Laville, P., Lehuger, S., Loubet, B., Chaumartin, F., Cellier, P., 2011. Effect of management, climate and soil conditions on N2O and NO emissions from an arable crop rotation using high temporal resolution measurements. Agr. Forest. Meteorol. 151, 228–240 doi: 10.1016/j.agrformet.2010.10.008.

Lehuger, S., Gabrielle, B., van Oijen, M., Makowski, D., Germon, J.-C., Morvan, T., Hénault, C., 2009. Bayesian calibration of the nitrous oxide emission module of an agro-ecosystem model. Agr. Ecosys. Environ. 133, 208–222 doi: 10.1016/j.agee.2009.04.022.

Li, C., Frolking, S., Harriss, R., 1994. Modeling carbon biogeochemistry in agricultural soils. Global Biogeochem. Cycles 8, 237–254 doi: 10.1029/94GB00767.

Li, C., Narayanan, V., Harriss, R.C., 1996. Model estimates of nitrous oxide emissions from agricultural lands in the United States. Global Biogeochem. Cycles 10, 297–306 doi: 10.1029/96GB00470.

Liang, C., MacDonald, D., Thiagarajan, A., Flemming, C., Cerkowniak, D., Desjardins, R., 2020. Developing a country specific method for estimating nitrous oxide emissions from agricultural soils in Canada. Nutr. Cycl. Agroecosystems 117, 145–167 doi: 10.1007/s10705-020-10058-w.

Little, S.M., Lindeman, J., Maclean, K., Janzen, H.H., 2008. Holos - A tool to estimate and reduce GHGs from farms. Methodology and algorythms for version 1.1. AAFC Agriculture and Agri-Food Canada, Ottawa.

Loubet, B., Laville, P., Lehuger, S., Larmanou, E., Fléchard, C., Mascher, N., Genermont, S., Roche, R., Ferrara, R.M., Stella, P., Personne, E., Durand, B., Decuq, C., Flura, D., Masson, S., Fanucci, O., Rampon, J.-N., Siemens, J., Kindler, R., Gabrielle, B., Schrumpf, M., Cellier, P., 2011. Carbon, nitrogen and Greenhouse gases budgets over a four years crop rotation in northern France. Plant Soil 343, 109–137 doi: 10.1007/s11104-011-0751-9.

Macdiarmid, J.I., Douglas, F., Campbell, J., 2016. Eating like there's no tomorrow: Public awareness of the environmental impact of food and reluctance to eat less meat as part of a sustainable diet. Appetite 96, 487–493 doi: 10.1016/j.appet.2015.10.011.

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). Anim. 12, 383–397 doi: 10.1017/S1751731117001847.

Macombe, C., Loeillet, D., Gillet, C., 2018. Extended community of peers and robustness of social LCA. Int. J. Life Cycle Assess. 23, 492–506 doi: 10.1007/s11367-016-1226-2.

Marton, S.M.R.R., Zimmermann, A., Kreuzer, M., Gaillard, G., 2016. Comparing the environmental performance of mixed and specialised dairy farms: the role of the system level analysed. J. Clean. Prod. 124, 73–83 doi: 10.1016/j.jclepro.2016.02.074.

McConkey, B., Ogle, S.M., Chirinda, N., Kishimoto-Mo, A.W., Baldock, J., Trunov, A., Alsaker, C., Lehmann, J., Woolf, D. (Eds,), 2019. Chapter 6: Grassland, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC, Intergovernmental panel for climate change, Geneva, Switzerland.

Metivier, K.A., Pattey, E., Grant, R.F., 2009. Using the ecosys mathematical model to simulate temporal variability of nitrous oxide emissions from a fertilized agricultural soil. Soil Biol. Biochem. 41, 2370–2386 doi: 10.1016/j.soilbio.2009.03.007.

Morris, J., Brown, S., Cotton, M., Matthews, H.S., 2017. Life-Cycle Assessment Harmonization and Soil Science Ranking Results on Food-Waste Management Methods. Environ. Sci. Technol. 51, 5360–5367 doi: 10.1021/acs.est.6b06115.

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 doi: 10.1016/j.eja.2015.01.005.

Nemecek, T., Roesch, A., Bystricky, M., Jeanneret, P., Lansche, J., Stüssi, M., Gaillard, G., 2023. Swiss Agricultural Life Cycle Assessment: A method to assess the emissions and environmental impacts of agricultural systems and products. Int. J. Life Cycle Assess. doi: 10.1007/s11367-023-02255-w.

Oertel, C., Matschullat, J., Zurba, K., Zimmermann, F., Erasmi, S., 2016. Greenhouse gas emissions from soils—A review. Geochemistry 76, 327–352 doi: 10.1016/j.chemer.2016.04.002.

Ogle, S., Wakelin, S.J., Buendia, L., McConckey, B., Baldock, J., Akiyama, H., Kishimoto-Mo, A.M., Chirinda, N., Bernoux, M., Bhattacharya, S., Chuersuwan, N., Goheer, M.A.R., Hergoualc'h, K., Ishizuka, S., Lasco, R.D., Pan, X., Pathak, H., Regina, K., Sato, A., Vazquez-Amabile, G., Wang, C., Zheng, X., Alsaker, C., Cardinael, R., Corre, M.D., Gurung, R., Mori, A., Lehmann, J., Rossi, S., Van Straaten, O., Veldkamp, E., Woolf, d., Yagi, K., Yan, X., 2019a. Chapter 5: Cropland, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, IPCC International Panel on Climate Change, Geneva.

Ogle, S., Kurz, W.A., Green, C., Brandon, A., Baldock, J., Domke, J., Herold, M., Bernoux, M., Chirinda, N., De Ligt, R., Federici, S., Garcia, E., Grassi, G., Gschwantner, T., Hirata, Y., Houghton, R., House, J.J., Ishizuka, S., Jonckheere, I., Krisnawati, H., Lehtonen, A., Kinyanjui, M.J., McConkey, B., Naesset, E., Niinistö, S.M., Ometto, J.P., Panichelli, L., Paul, T., Peterson, H., Reddy, S., Regina, K., Rocha, M., Rock, J., Sanz-Sanchez, M., Sanquetta, S., Sato, S., Somogyi, Z., Trunov, A., Vazquez-Amabile, G., Vitullo, M., Wang, C., Waterworth, R.M., Collet, M., Harmon, M., Lehmann, J., Shaw, C.H., Shirato, Y., Wolf, D., 2019b. Chapter 2: Generic methodologies, applicable to multiple land-use categories, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, IPCC, Intergovernmental panel for climate change, Geneva.

Olesen, J.E., Rees, R.M., Recous, S., Bleken, M.A., Abalos, D., Ahuja, I., Butterbach-Bahl, K., Carozzi, M., De Notaris, C., Ernfors, M., Haas, E., Hansen, S., Janz, B., Lashermes, G., Massad, R.S., Petersen, S.O., Rittl, T.F., Scheer, C., Smith, K.E., Thiébeau, P., Taghizadeh-Toosi, A., Thorman, R.E., Topp,

C.F.E., 2023. Challenges of accounting nitrous oxide emissions from agricultural crop residues. Glob. Change Biol. gcb.16962 doi: 10.1111/gcb.16962.
Parajuli, R., Dalgaard, T., Birkved, M., 2018. Can farmers mitigate environmental impacts through combined production of food, fuel and feed? A consequential life cycle assessment of integrated mixed crop-livestock system with a green biorefinery. Sci. Tot. Environ. 619–620, 127–143 doi: 10.1016/j.scitotenv.2017.11.082.
Parton, W.J., Ojima, D.S., Cole, C.V., Schimel, D.S., 1994. A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. Pages 147-167, in: Bryant, in R.B., R. W. 'Arnold, editors. (Eds.), Quantitative modeling of soil forming processes. Soil Science Society of America, Madison, Wisconsin, USA.
Pattey, E., Edwards, G., Desjardins, R., Pennock, D., Smith, W., Grant, B., Macpherson, J., 2007. Tools for quantifying N2O emissions from agroecosystems. Agr. Forest Met. 142, 103–119 doi: 10.1016/j.agrformet.2006.05.013.
Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Sci. 360, 987–992 doi: 10.1126/science.aaq0216.
Pörtner, H.-O., Roberts, D.C., Adams, H., Adelekan, I., Adler, C., Adrian, R., Aldunce, P., Ali, E., Begum, R.A., BednarFriedl, B., Kerr, R.B., Biesbroek, R., Birkmann, J., Bowen, K., Caretta, M.A., Carnicer, J., Castellanos, E., Cheong, T.S., Chow, W., Cissé, G., Clayton, S., Constable, A., Cooley, S.R., Costello, M.J., Craig, M., Cramer, W., Dawson, R., Dodman, D., Efitre, J., Garschagen, M., Gilmore, E.A., Glavovic, B.C., Gutzler, D., Haasnoot, M., Harper, S., Hasegawa, T., Hayward, B., Hicke, J.A., Hirabayashi, Y., Huang, C., Kalaba, K., Kiessling, W., Kitoh, A., Lasco, R., Lawrence, J., Lemos, M.F., Lempert, R., Lennard, C., Ley, D., Lissner, T., Liu, Q., Liwenga, E., Lluch-Cota, S., Löschke, S., Lucatello, S., Luo, Y., Mackey, B., Mintenbeck, K., Mirzabaev, A., Möller, V., Vale, M.M., Morecroft, M.D., Mortsch, L., Mukherji, A., Mustonen, T., Mycoo, M., Nalau, J., New, M., Okem, A., Ometto, J.P., O'Neill, B., Pandey, R., Parmesan, C., Pelling, M., Pinho, P.F., Pinnegar, J., Poloczanska, E.S., Prakash, A., Preston, B., Racault, M.-F., Reckien, D., Revi, A., Rose, S.K., Schipper, E.L.F., Schmidt, D.N., Schoeman, D., Shaw, R., Simpson, N.P., Singh, C., Solecki, W., Stringer, L., Totin, E., Trisos, C.H., Trisurat, Y., Aalst, M., Viner, D., Wairiu, M., Warren, R., Wester, P., Wrathall, D., Ibrahim, Z.Z., 2022. Technical Summary. Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, in: Pörtner, H.-O., Roberts, D.C., Tignor, M., Poloczanska, E.S., Mintenbeck, K., Alegría, A., Craig, M., Langsdorf, S., Löschke, S., Möller, V., Okem, A., Rama, B. (Eds.), Cambridge University Press, Cambridge, UK and New York, NY, USA, pp. 37-118, doi: 10.1017/9781009325844.002.
Potting, J., Hauschild, M.Z., 2006. Spatial differentiation in life cycle impact assessment: A decade of method development to increase the environmental realism of LCIA. Int. J. Life Cycle Assess. 11, 11–13 doi: 10.1065/lca2006.04.005.

Rochette, P., Eriksen-Hamel, N.S., 2008. Chamber Measurements of Soil Nitrous Oxide Flux: Are Absolute Values Reliable? Soil Sci. Soc. Am. J. 72, 331–342 doi: 10.2136/sssaj2007.0215.

Rochette, P., Liang, C., Pelster, D., Bergeron, O., Lemke, R., Kroebel, R., MacDonald, D., Yan, W., Flemming, C., 2018. Soil nitrous oxide emissions from agricultural soils in Canada: Exploring relationships with soil, crop and climatic variables. Agr. Ecosys. Environ. 254, 69–81 doi: 10.1016/j.agee.2017.10.021.

Rotz, C.A., 2018. Modeling greenhouse gas emissions from dairy farms. J. Dairy Sci. 101, 6675–6690 doi: 10.3168/jds.2017-13272.

Saggar, S., 2010. Estimation of nitrous oxide emission from ecosystems and its mitigation technologies. Agr. Ecosyst. Environ. 136, 189–191 doi: 10.1016/j.agee.2010.01.007.

Schmidt Rivera, X.C., Bacenetti, J., Fusi, A., Niero, M., 2017. The influence of fertiliser and pesticide emissions model on life cycle assessment of agricultural products: The case of Danish and Italian barley. Sci. Tot. Environ. 592, 745–757 doi: 10.1016/j.scitotenv.2016.11.183.

Segura-Salazar, J., Lima, F.M., Tavares, L.M., 2019. Life Cycle Assessment in the minerals industry: Current practice, harmonization efforts, and potential improvement through the integration with process simulation. J. Clean. Prod. 232, 174–192 doi: 10.1016/j.jclepro.2019.05.318.

Siegert, M.-W., Lehmann, A., Emara, Y., Finkbeiner, M., 2019. Harmonized rules for future LCAs on pharmaceutical products and processes. Int. J. Life Cycle Assess. 24, 1040–1057 doi: 10.1007/s11367-018-1549-2.

Sieverding, H., Kebreab, E., Johnson, J.M.F., Xu, H., Wang, M., Grosso, S.J.D., Bruggeman, S., Stewart, C.E., Westhoff, S., Ristau, J., Kumar, S., Stone, J.J., 2020. A life cycle analysis (LCA) primer for the agricultural community. Agron. J. 112, 3788–3807 doi: 10.1002/agj2.20279.

Sinisterra-Solís, N.K., Sanjuán, N., Estruch, V., Clemente, G., 2020. Assessing the environmental impact of Spanish vineyards in Utiel-Requena PDO: The influence of farm management and on-field emission modelling. J. Environ. Manag. 262, 110325 doi: 10.1016/j.jenvman.2020.110325.

Sozanska, M., Skiba, U., Metcalfe, S., 2002. Developing an inventory of N2O emissions from British soils. Atmospheric Environ. 36, 987–998 doi: 10.1016/S1352-2310(01)00441-1.

Styles, D., Gibbons, J., Williams, A.P., Stichnothe, H., Chadwick, D.R., Healey, J.R., 2014. Cattle feed or bioenergy? Consequential life cycle assessment of biogas feedstock options on dairy farms. GCB Bioenergy 7, 1034–1049 doi: 10.1111/gcbb.12189.

Styles, D., Gibbons, J., Williams, A.P., Dauber, J., Stichnothe, H., Urban, B., Chadwick, D.R., Jones, D.L., 2015. Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation. GCB Bioenergy 7, 1305–1320 doi: 10.1111/gcbb.12246.

Sykes, A.J., Macleod, M., Eory, V., Rees, R.M., Payen, F., Myrgiotis, V., Williams, M., Sohi, S., Hillier, J., Moran, D., Manning, D.A.C., Goglio, P., Seghetta, M., Williams, A., Harris, J., Dondini, M., Walton, J., House, J., Smith, P., 2019. Characterising the biophysical, economic and social impacts of soil

carbon sequestration as a greenhouse gas removal technology. Glob. Change Biol. 0 doi: 10.1111/gcb.14844.
Taki, R., Wagner-Riddle, C., Parkin, G., Gordon, R., VanderZaag, A., 2019. Comparison of two gap-filling techniques for nitrous oxide fluxes from agricultural soil. Can. J. Soil. Sci. 99, 12–24 doi: 10.1139/cjss-2018-0041.
Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Comparing energy balances, greenhouse gas balances and biodiversity impacts of contrasting farming systems with alternative land uses. Agr. Syst. 108, 42–49 doi: 10.1016/j.agsy.2012.01.004.
UNEP, 2023a.The Global LCA Data Access network (GLAD). http://www.unep.org/explore-topics/resource-efficiency/what-we-do/life-cycle-initiative/global-lca-data-access-network (accessed 20 december 2023).
UNEP, 2023b.Global Guidance on Environmental Life Cycle Impact Assessment Indicators (GLAM). https://eplca.jrc.ec.europa.eu/glam.html (accessed 20 december 2023).
Ussiri, D., Lal, R., 2013. Formation and Release of Nitrous Oxide from Terrestrial and Aquatic Ecosystems, in: Ussiri, D., Lal, R. (Eds.), Soil Emission of Nitrous Oxide and Its Mitigation. Springer Netherlands, Dordrecht, pp. 63–96 doi: 10.1007/978-94-007-5364-8_3.
Ussiri, D.A.N., Lal, R., Jarecki, M.K., 2009. Nitrous oxide and methane emissions from long-term tillage under a continuous corn cropping system in Ohio. Soil Till. Res. 104, 247–255 doi: 10.1016/j.still.2009.03.001.
van der Werf, H.M.G., Knudsen, M.T., Cederberg, C., 2020. Towards better representation of organic agriculture in life cycle assessment. Nat. Sustain. doi: 10.1038/s41893-020-0489-6.
Van Zanten, H.H.E., Herrero, M., Van Hal, O., Röös, E., Muller, A., Garnett, T., Gerber, P.J., Schader, C., De Boer, I.J.M., 2018. Defining a land boundary for sustainable livestock consumption. Glob. Change Biol. 24, 4185–4194 doi: 10.1111/gcb.14321.
Venterea, R.T., Bijesh, M., Dolan, M.S., 2011. Fertilizer Source and Tillage Effects on Yield-Scaled Nitrous Oxide Emissions in a Corn Cropping System. J. Environ. Qual. 40, 1521 doi: 10.2134/jeq2011.0039.
Venterea, R.T., Petersen, S.O., de Klein, C.A.M., Pedersen, A.R., Noble, A.D.L., Rees, R.M., Gamble, J.D., Parkin, T.B., 2020. Global Research Alliance N2O chamber methodology guidelines: Flux calculations. J. Environ. Qual. 49, 1141–1155 doi: https://doi.org/10.1002/jeq2.20118.
Wang, Y., Guo, J., Vogt, R.D., Mulder, J., Wang, J., Zhang, X., 2018. Soil pH as the chief modifier for regional nitrous oxide emissions: New evidence and implications for global estimates and mitigation. Glob. Change Biol. 24, e617–e626 doi: https://doi.org/10.1111/gcb.13966.
Welegedara, N.P.Y., Grant, R.F., Quideau, S.A., Das Gupta, S., 2020a. Modelling nitrogen mineralization and plant nitrogen uptake as affected by reclamation cover depth in reclaimed upland forestlands of Northern Alberta. Biogeochemistry 149, 293–315 doi: 10.1007/s10533-020-00676-5.

Welegedara, N.P.Y., Grant, R.F., Quideau, S.A., Landhäusser, S.M., Merlin, M., Lloret, E., 2020b. Modelling plant water relations and net primary productivity as affected by reclamation cover depth in reclaimed forestlands of northern Alberta. Plant Soil 446, 627–654 doi: 10.1007/s11104-019-04363-9.
Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Garnett, T., Tilman, D., Wood, A., DeClerck, F., Jonell, M., Clark, M., Gordon, L., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J.A., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S., Reddy, S., Narain, S., Nishtar, S., Murray, C., 2019. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. The Lancet 393, 447–492.
Zaher, U., Stöckle, C., Painter, K., Higgins, S., 2013. Life cycle assessment of the potential carbon credit from no- and reduced-tillage winter wheat-based cropping systems in Eastern Washington State. Agric. Syst. 122, 73–78 doi: 10.1016/j.agsy.2013.08.004.
Zampori, L., Pant, R., 2019.Suggestions for updating the Product Environmental Footprint (PEF) method. EUR 29682 EN, Publications Office of the European Union, Luxembourg.
Zhou, J., Zheng, Y., Hou, L., An, Z., Chen, F., Liu, B., Wu, L., Qi, L., Dong, H., Han, P., Yin, G., Liang, X., Yang, Y., Li, X., Gao, D., Li, Y., Liu, Z., Bellerby, R., Liu, M., 2023. Effects of acidification on nitrification and associated nitrous oxide emission in estuarine and coastal waters. Nat. Comm. 14, 1380 doi: 10.1038/s41467-023-37104-9.

SG5c: Evaluating manure impact methodologies within Life Cycle Assessments of livestock systems and products

Pietro Goglio[a], Simon Moakes[b,c], Mariano Pauselli[a], Emanuele Lilli[a], Manuel Romero-Huelva[j], Muhammad Ahmed Waqas[d,e], David Yanez-Ruiz[j], Laurence G. Smith[g, k]

[a]Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy
[b]Department of Food System Sciences, Research Institute of Organic Agriculture (FiBL), Frick, Switzerland
[c]IBERS, Aberystwyth University, UK
[g]School of Agriculture, Policy and Development, University of Reading, UK
[j]Estación Experimental del Zaidín (CSIC), Profesor Albareda 1, 18008 Granada, Spain
[k]Department of Biosystems and Technology, Swedish University of Agricultural Sciences, Box 190, SE-234 22 Lomma, Sweden

Abstract

Worldwide greenhouse gas emissions (GHG) reached 59 Gt of CO_2 eq. in 2019 with an increasing demand for livestock sector GHG emissions to be reduced. LCA has been successful in assessing GHGs from livestock systems. However, no harmonization attempt has been carried out, despite the need to improve LCA methodologies for assessing GHG in the LCA of livestock systems. We therefore undertook a review of existing manure (storage and housing) assessment methods as part of an effort to develop a coherent harmonisation approach for livestock LCA. The approach adopted was based on two anonymous expert surveys and a series of expert workshops (n=21) to define general and specific criteria to review LCA methods for GHG emissions used in LCA of livestock systems. More than 29,151 scientific papers and reports were identified, 1175 were screened and 48 included in the final manure and housing GHG review. The results showed that a high level of accuracy corresponded to a low level of applicability and vice versa. Thus, the choice of the methodology in relation to the LCA objectives is a particularly critical for a high quality LCA assessment. Following the analysis of the available literature, a series of recommendations were proposed. Whilst IPCC Tier 1 methodology has been employed in most of the assessments analysed, the more detailed Tier 2 methods, related to the specifics of the manure and housing systems are preferable for improved accuracy. Furthermore, as a general recommendation for estimating the GHG from livestock systems, the choice of LCA

methods should be based on the LCA objectives, data availability and expertise of the LCA practitioner. Future development of LCA methodologies is necessary to improve LCAs of livestock systems. This LCA method development should be synchronous with improvements of observation methods and the assessment of different crop-livestock management.

Introduction

Worldwide greenhouse gas emissions reached 59 Gt of CO_2eq in 2019 with the agriculture forestry and land use sector contributing around 22% of total emissions. Thus, there is an increasing demand for greenhouse emission reduction for every sector of the economy, including agriculture (IPCC, 2022). At the same time, worldwide demand for animal products is predicted to double over the next decades due to population growth and increasing economic prosperity (Godfray et al., 2018).

It is estimated that livestock supply chains are responsible for 14.5% of all anthropogenic greenhouse gas emissions (FAO, 2017). Within the sector, feed production, manure management and enteric fermentation are the main contributors to climate change impacts. N_2O and CH_4 emissions from manure management contributed 4.3% and 5.7% to global greenhouse gas emissions of livestock production chains respectively while CH_4 from enteric fermentation accounted for 44.1% of the total livestock emissions (FAO, 2017). N_2O emissions from the application and deposition of manure and nitrous oxide emissions from fertilizers and crop residues in feed production contributed for 13.4% and 5.8% to the livestock sector's emissions respectively, while CO_2 emissions from feed production contributed 13% (FAO, 2017). In addition to the GHG emissions, soil contains the largest share of terrestrial carbon under a dynamic equilibrium which depends *inter alia* on soil types, climate, and management practices.

Accounting for fluxes of CO_2 and N_2O in agro-ecosystems is important for evaluating the enhancing or mitigating climate change effects of different livestock systems (Grossi et al., 2019; Sykes et al., 2019). In general, CO_2 is mainly released from soils as a product of microbial or root respiration (Lal and Stewart, 2018), whereas some of the CO_2 that has been removed from the atmosphere through photosynthesis can be sequestered as carbon in soil organic matter (Oertel et al., 2016b; Paustian et al., 2016). Soil CO_2 and N_2O emissions from soils are evaluated mostly with regards to land management changes (e.g. tillage, fertilisation) and land use changes (from and to grassland/ cropland/ forest), following intergovernmental panel for climate change (IPCC) classification (McConkey et al., 2019; Ogle et al., 2019a, 2019b).

Manure handling and storage are both associated with GHG emissions. Emissions related to manure handling are largely affected by the type of storage (Owen and Silver, 2015), including factors such as the formation of a superficial crust (Owen and Silver, 2015). The crust favours the presence of aerobic and anaerobic microsites which cause nitrification and denitrification leading to N_2O release (Philippe and Nicks, 2015). In the case of slurry, temperature, N content, solid content are the main drivers to

N_2O emissions (Brady and Weil, 2002; Gavrilova et al., 2019). Slurry is often treated by anaerobic digesters, which can increase methane losses due to leakage which is also an important factor affecting emissions (FAO, 2016a; Rotz, 2018). Manure emissions whilst housed are also responsible for GHG emissions attributed to livestock systems and products, which is affected by temperature, ventilation, floor type, feed composition, manure/ removal strategy and type of bedding (Bohran et al., 2012; FAO, 2016a; Philippe and Nicks, 2015).

Enteric fermentation emissions are generated in the digestive system of livestock during the fermentation of feed. Whilst monogastrics produce minimal methane emissions per animal, ruminants generate far greater quantities per animal due to processes within the rumen as fibre is broken down. The process generates, inter alia, hydrogen, carbon dioxide and methane. In particular, the amount of methane released depends on many aspects, such as the type of digestive tract, the age and weight of the animal, and the type and quantity of the feed consumed (Gavrilova et al., 2019).

Life Cycle Assessment (LCA) is commonly used to assess livestock systems and products due to its ability to identify environmental hotspot and trade-offs across different types of pollution (Cederberg et al., 2013). LCA has been also widely used to assess climate change impacts of food and livestock products (Grossi et al., 2019; Poore and Nemecek, 2018). It has been widely utilised to assess livestock systems including pig production (McAuliffe et al., 2016), beef (Flysjö et al., 2012; Peters et al., 2010), milk and dairy systems including cheese production (Flysjö et al., 2012; Kim et al., 2013; Kristensen et al., 2015), sheep and lamb production system Life Cycle assessment (LCA) is an assessment method commonly used to assess livestock systems and products due to its ability to identify environmental hotspot and trade-offs across different types of pollution (Cederberg et al., 2013). t has been widely utilised to assess livestock systems including pig production (McAuliffe et al., 2016), beef (Flysjö et al., 2012; Peters et al., 2010), milk and dairy systems including cheese production (Flysjö et al., 2012; Kim et al., 2013; Kristensen et al., 2015), sheep and lamb production systems (Bhatt and Abbassi, 2021; Vagnoni et al., 2015), and poultry production systems (Kalhor et al., 2016; López-Andrés et al., 2018; Skunca et al., 2018b; Williams et al., 2016).

Several harmonisation attempts were carried out in sectors other than agriculture (Segura-Salazar et al., 2019; Siegert et al., 2019), while others focused on wines (Jourdaine et al., 2020) or food waste advocating for a better integration between life cycle assessment and soil science (Morris et al., 2017) (Morris et al., 2017). No harmonization attempt has been made for soil C, soil N_2O emissions, manure emissions and enteric fermentation. Although, recent guidelines have been proposed by the Food and Agriculture Organisation (FAO, 2016a, 2016b, 2016c, 2016d, 2016e, 2020). However, these reports are mostly prescriptive (i.e.. suggesting methodology) and they contain a limited comparison and discussion of methods.

Within this study, we undertook a coherent harmonisation approach for GHG to assess LCA methods in livestock systems and production chains focused on: i) soil CO_2 emissions related to livestock chains; ii) soil N2O emissions, iii) manure emissions (i.e.. storage, housing); iv) enteric fermentation.

For the purpose of this report, manure spreading GHG emissions are considered as part of the soil field emissions and will be dealt as part of the soil CO_2 and N_2O emissions, as previously carried out in LCA research (Goglio et al., 2017; Petersen et al., 2013). The overall aim of this work is to present advantages or disadvantages of the LCA methods reviewed to inform LCA practitioners and researchers to identify future methodological research needs.

Methodology

SEARCH, SCREENING CRITERIA, DATA PROCESSING

Search criteria

A literature search was conducted using Scopus, Elsevier, Google Scholar and Web of science search engines. The search terms and search term combinations employed were described below in table 1 and included all papers published between 2012 to 2022.

Screening and review procedures

The collected sources were screened against the following criteria:

- Peer-reviewed publication in a scientific journal or published by FAO, European Commission
- English language publication
- Method is related to and applicable for LCA
- Method is related to livestock systems or its components
- Method is applicable for European livestock systems

A systematic review of the existing literature, based on the methodology described above was conducted to provide a comprehensive assessment on how LCA methodologies include the issue of livestock GHG emissions related to soil CO_2, soil N_2O, manure (housing and storage) and enteric fermentation. This critical review sought to identify the most significant components and summarize the main concepts. To achieve this, a review protocol was developed (Table 1), describing the search and screening process including an iterative process of article selection based on restrictive criteria. First ("*identification step*") the literature search was performed, according to the queries defined in Table 1, in Scopus, Web of Science and Google Scholar databases. Searches led to a total of 29 151 papers. Only articles published during the 2012-2022 period in the following research areas: Agriculture; Agriculture or Soil or Animals or Cattle or Dairying or Crop production or Animal feed or Animal Husbandry or Swine or Livestock or Chickens or Poultry. When the Google search engine was used in the search, the selection of papers was stopped at page 15 of the search results. Papers with research not fully relevant for the livestock sector such as rice, plastic, biofuel, bioenergy were excluded. Energy papers related to biogas without any relation to feed, soil emissions were also excluded as for papers with insects, fish or feed production without any focus on livestock.

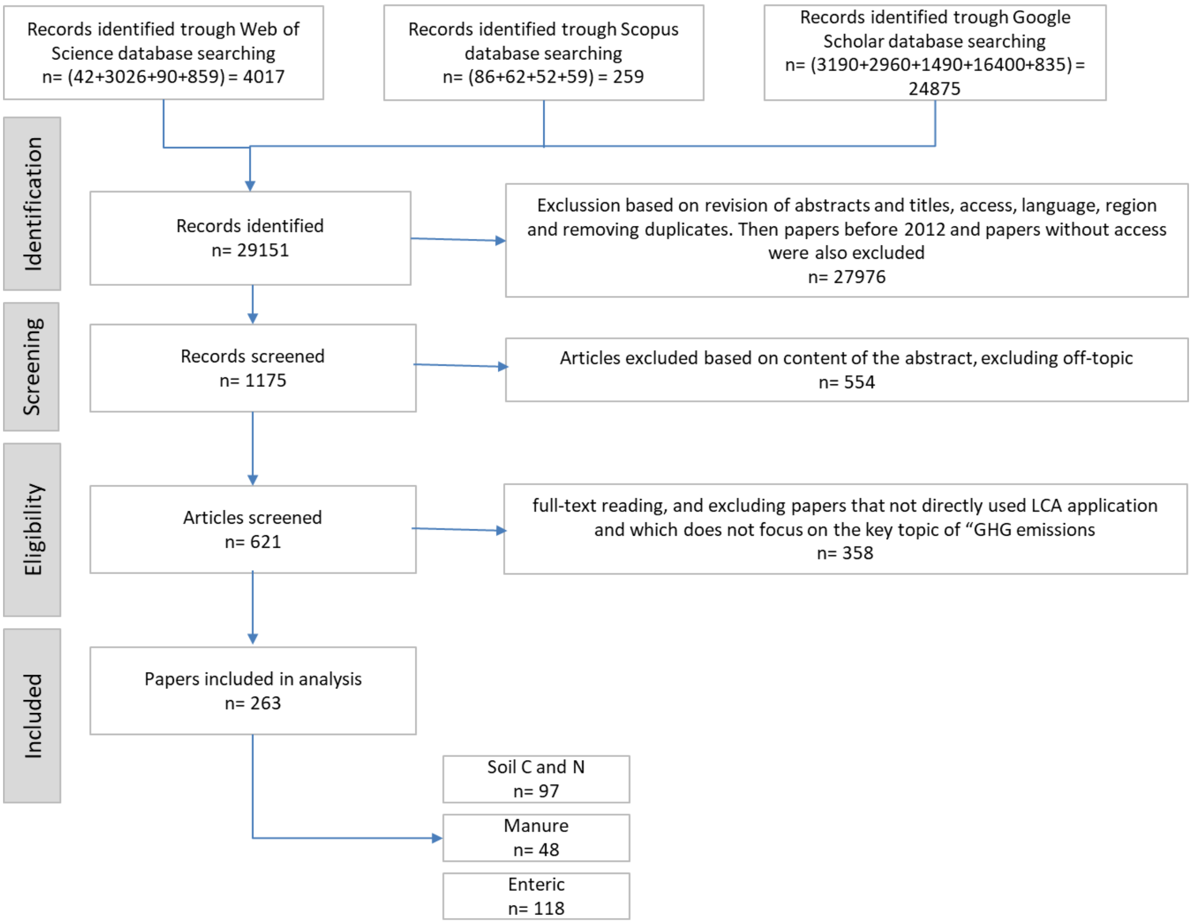
Table 1 - Combinations of search terms for the subgroup "GHG Emission Issues"

Database	Combination	Search strings[1]
Scopus & Web of Science	1	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("enteric fermentation")
	2	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("soil*") AND ("emissions" OR "nitrous oxide" OR "N2O" OR "carbon dioxide" OR "CO2" OR "carbon sequestration" OR "GHG" OR "greenhouse gas*" OR "C dynamics" OR "soil) AND ("carbon") AND ("livestock")
	3	("Life Cycle Assessment" OR " life cycle analysis") AND ("wheat" OR "maize" OR "grass" OR "barley" OR "oat" OR "soy*" OR "faba beans" OR "alfalfa" OR "clover" OR "sorghum" OR "Rye" OR "Ley") AND ("soil emissions" OR "soil carbon" OR "soil nitrogen" OR "soil organic matter" OR "nitrous oxide") AND ("feed" OR "fodder" OR "farming system" OR "farm")
	4	("Life Cycle Assessment" OR " life cycle analysis") AND ("livestock" OR "dairy" OR "cattle" OR "sheep" OR "pig*" OR "poultry" OR "goat*" OR "milk" OR "egg*" OR "chicken*" OR "cow*" OR "husbandry") AND ("emissions") NOT ("waste" OR "biofuel" OR "bioenergy")
	5	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("manure" OR "slurry") AND ("handling" OR "storage" OR "treatment" OR "emissions")
	6	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("emissions") AND ("livestock*" OR "dairy" OR "sheep" OR "pig" OR "poultry" OR "goat" OR "milk" OR "egg*" OR "Chicken" OR "cow "NOT "waste" OR "biofuel" OR "bioenergy")
Google Scholar	7	"LCA" "enteric fermentation" OR "enteric emissions"
	8	"LCA" "manure application" OR "manure emissions"
	10	"LCA" "crop soil emissions"
	11	"LCA" "livestock"
	18	"LCA" "wheat soil emissions"

[1]Last accessed in March 2022

Reviewing abstracts and titles, access, language, region and then removing duplicate papers. The second step of "Screening" was made by using restrictive criteria ("refine results") excluding appearances before 2012 and papers without access (1.175 papers), and a second selection was performed based on the content of the abstract, excluding off-topic. Finally, 621 papers were selected as "Eligible" for a full-text reading. The last step, following the full-text reading, excluded papers that not directly used in the LCA application or did not focus on the key topic of "GHG emissions". Through this iterative process, the total amount of papers included in the qualitative analysis was reduced to 263 papers, and specifically, 48 related to manure and housing GHG.

Figure 1 - Methodological steps of the literature search



GENERAL CRITERIA SELECTION

The papers identified as part of this harmonization were then reviewed using both general criteria and specific criteria to assess the LCA methods for livestock systems and products. General criteria used in the harmonization of LCA methods for livestock systems for GHG emissions were selected using a participatory approach based on a modified DELPHI method, as previously extensively described and here briefly summarized (Goglio et al., 2023). The selection of key topics was carried out through an anonymous survey which allowed us to screen the various topics and provide a priority list on the basis of a preliminary literature review with key words: "life cycle assessment", "livestock", "poultry", "beef", "dairy", "milk", "cheese", "meat", "pig", "pork", "turkey", "sheep", "lamb" and "goat", "methods", "harmonisation", "review", "methodology". Within the survey each participant was invited to express a priority value with a range from one (low priority) to ten (high priority). On this basis, specific criteria and review approaches were developed for each key topic.
A review of frameworks used to assess LCA methods was undertaken, and key search words included: "LCA methods", "LCA framework", "livestock", "agriculture". Articles and publications were collected from literature including the FAO LEAP reports and the PEFCR general guidelines (FAO, 2018; Zampori and Pant, 2019). Only publicly available documents were screened.
An anonymous survey was carried out using a Google survey (Google, 2024), involving LCA experts. The general criteria selected through the survey were then further screened through LCA expert discussions to ensure that both the definition and the scale would be coherent with the harmonization efforts of the LCA methodology for livestock systems and products. For some criteria the definition and the scale was reformulated and modified to ensure rigour and coherence in the review of the LCA methods (Goglio et al., 2023). Table 2 presented the set of general criteria defined for the harmonization of LCA methods for livestock systems.

SPECIFIC CRITERIA IDENTIFICATION

Following the definition of the general criteria, specific evaluation criteria were defined for each specific topic in several workshops (n=4). The definition and the scale of specific criteria were reformulated and modified to ensure rigour and coherence in the review of the LCA methods. The expert discussions were conducted as a community of peers (Macombe et al., 2018) and different specific criteria were selected for each key area.
The following key areas were selected to be investigated with regards to the livestock LCA assessment methodology:

- Manure management (storage and treatment) (CO_2, CH_4 and N_2O emissions)

To undertake this a systematic literature review was undertaken to assess how GHG emissions from soil, from manure handling and methane emissions from enteric fermentation are evaluated within the LCA of livestock systems. Each sub-topic is subsequently described.

Table 2 - Matrix of general criteria description and the correspondent scale used for the critical review of LCA methodologies.

General criteria and definition	Level 1	Level 2	Level 3	Level 4
Transparency and reproducibility: Comprehensive documentation and mechanisms that allow reviewers to verify/review all data, calculations, and assumptions	LCA methodologies which do not allow reviewers to verify/review the results, calculations and assumptions.	LCA methodologies which could be reviewed together with the results but some calculations and assumptions cannot be reviewed.	LCA methodologies which fully allows reviewers to verify/review the results, calculations and assumptions	
Completeness: Relationships between quantification of the environmental impact (material/energy flows and other environmental interventions) and adherence to the defined system boundary, the data requirements, and the impact assessment methods employed	The quantification of the environmental impacts including all material/energy flows and other environmental interventions do not have adherence to the system boundary, the data requirements and the impact assessment methods employed	The quantification of the environmental impacts is conform either to the defined system boundary or the data requirements or the system method employed;	The quantification of the environmental impacts conforms to two aspects between the defined system boundary, data requirements and impact assessment method employed.	The quantification of the environmental impacts fully corresponds to the system boundary, data requirements and the impact assessment methods employed.
Fairness and acceptance: Level playing field across competing products, processes and industries	The LCA methodology does not provide level playing field across products, processes and industries;	The LCA methodology provides a level playing field for at least two products, processes, and industries (e.g., Beef and diary, beef and pig);	LCA provides a level playing field for several products, processes and industries.	
Robustness: Associated in the RACER framework the following sub criteria of providing a defensible theory, Sensitivity, Data quality, Reliability, Consistency, Comparability, Boundaries	The LCA methodology is not based on defensible theory, lacks sensitivity on certain environmental impacts either because of its reliability, comparability, the chosen system boundary or its comparability.	The LCA methodology is based on a defensible theory but it lacks sensitivity, reliability, comparability and it is not in agreement with the system boundaries.	The LCA methodology is based on a defensible theory with a satisfactory sensitivity, reliability, data quality, consistency, comparability and in agreement with the system boundaries.	

Applicability: Ability of the method to be used by a wide range of LCA practitioners	The LCA method can only be used with LCA expertise and extensive data availability	The LCA method can be used with either limited LCA expertise or data availability	The LCA method can be easily used with very limited LCA expertise and data availability

MANURE MANAGEMENT (HOUSING AND STORAGE)

Both manure handling and storage result in GHG emissions. For manure, GHGs originate from three biochemical processes: (1) urea hydrolysis producing CO_2 (2) anaerobic fermentation of organic matter into intermediate volatile fatty acids (VFAs), CH_4 and CO_2; (3) aerobic degradation of organic matter, producing CO_2 and ammonium N (Philippe and Nicks, 2015, Schlegel 2000), which can be converted in NO_3^-, a precursor of N_2O (Brady and Weil 2002, Schlegel 2000). For solid manure, GHGs can also be generated via anaerobic degradation producing mainly CH_4 and NH_3, a precursor of indirect N_2O emissions. However, GHGs from manure can be also produced via aerobic production realising CO_2 and ammonium N, a precursor of N_2O, through composting, which is influenced by several drivers including temperature, moisture content, C/N ratio, degradability of carbon compounds, pH level and the physical structure of the organic biomass (Gavrilova et al., 2019; Philippe and Nicks, 2015; Rotz, 2018, Brady and Weil 2002, Schlegel 2000).
Emissions related to manure handling are largely affected by the type of storage (Owen and Silver, 2015). A particular factor affecting the overall methane and N_2O emissions in manure handling is the formation of a superficial crust affecting N_2O release (Philippe and Nicks, 2015). In the case of slurry, temperature, N content and solid content are the main drivers to N_2O emissions (Brady and Weil, 2002; Gavrilova et al., 2019).
Housing is also another process responsible for GHG emissions, which is affected by temperature, ventilation, floor type, feed composition, manure/ removal strategy and type of bedding (Bohran et al., 2012; Philippe and Nicks, 2015). In anaerobic digesters, the level of methane leakage is also an important factor affecting emissions (FAO, 2016b).
Emissions related to manure application are not included in this "Manure management (handling and storage)" sub-topic, as they are included in the "Soil C dynamics & Soil N_2O emissions" sub-topic. The specific criteria selected for *"Manure management (handling and storage)"* are reported in Table 3.

Table 3 - Specific criteria to evaluate LCA methods for livestock systems and product related to manure management (housing and storage)

Specific Criteria description	Level 1	Level 2	Level 3	Level 4
Leakage inclusion for anaerobic digestion: These specific criteria can only be applied for livestock systems where anaerobic digestion is utilised. The leakage inclusion for anaerobic digestion criteria indicates whether methane leakage losses have been considered (FAO, 2016b; Grossi et al., 2019).	the methane leakage is not accounted for in the LCA method	The methane leakage is accounted for in the LCA method		
Accuracy in GHG of manure storage and treatment: The accuracy specific criteria for manure storage and treatment defines how much the LCA method is capable of capturing GHG emission drivers in manure handling and treatment which include temperature, moisture content, C/N ratio and pH variability, the crust formation (Philippe & Nicks, 2015). The proposed scale assumes that the LCA practitioner has sufficient expertise to adopt the methodology and that observations have been carried out with a protocol which aims at the maximum accuracy in observations, as previously discussed (FAO, 2018; Goglio et al., 2015)	the LCA method accounts only for the amount of manure produced;	the LCA method uses a default emission factor to the amount of manure produced;	the LCA method considers a dependency between the estimated GHG emissions and temperature, moisture content, C/N ratio, degradability of carbon compounds present in the manure, pH using average data;	the LCA method considers temperature, moisture content, C/N ratio, degradability of carbon compounds, pH with a daily time step or direct GHG measurements
Accuracy in GHG emissions due to animal housing: The accuracy in GHG emissions due to animal housing is a specific criteria which aims at assessing how key drivers affecting manure emissions due to housing are captured by the LCA methods. This scale assumes that the LCA practitioner has sufficient expertise to adopt the methodology and that observations have been carried out with a protocol which aims at the maximum accuracy in observations, as previously discussed (FAO, 2018; Goglio et al., 2015)	the LCA method accounts for manure emissions due to housing using standard emission factors;	the LCA method is based on a mechanistic model which considers temperature, ventilation, floor type, feed composition, manure/ removal strategy and type of bedding;	the LCA method is based on a model, which is affected by temperature using daily data, ventilation, floor type, feed composition, manure/ removal strategy and type of bedding; manure emissions;	the LCA methods includes daily monitoring of GHG emissions

![Pathways for Sustainable Food logo]

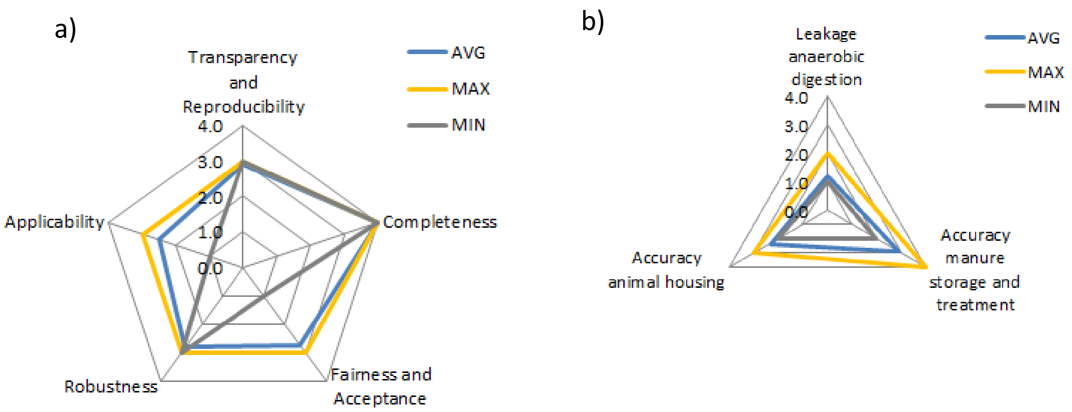
Results

QUANTITATIVE RESULTS

Manure emissions (Housing and storage)

For the manure emissions, the average scores for the general criteria assigned to papers are shown in Figure 2(a), the general criteria with the highest score were Completeness followed by Transparency and Reproducibility, Robustness, Fairness and Acceptance and finally Applicability. Figure 2a) also shows that, on average, papers received a score above or equal to the midpoint for each general criteria (i.e., 2). It is important to note that the general criteria have scoring scales with different maximum values (3 for all except Completeness).

Figure 2 - General Criteria Average Scores (a), Specific Criteria Scores (b) for manure emissions (housing and storage)

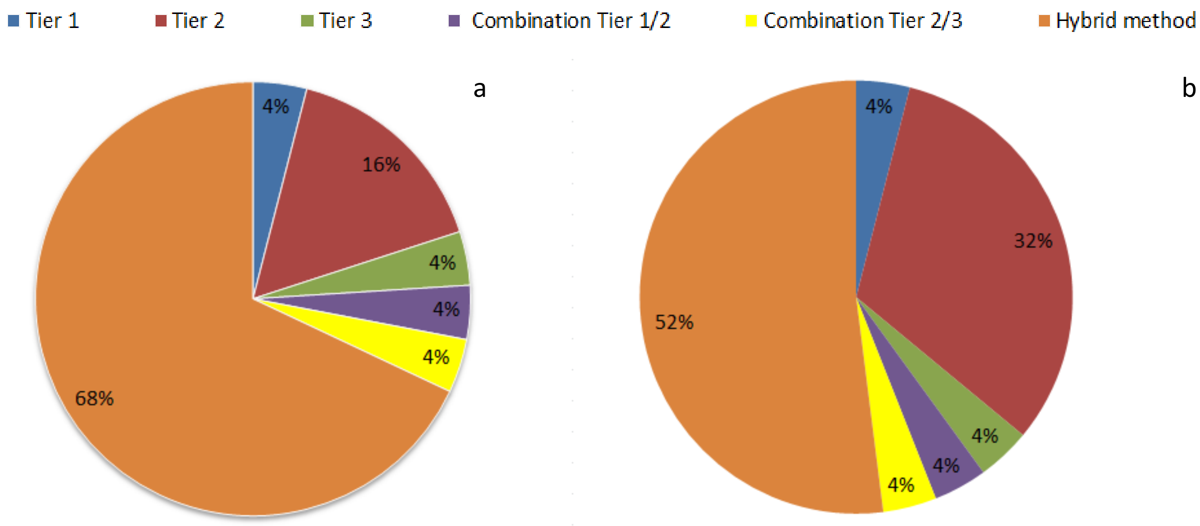


Specific criteria for manure emissions were leakage inclusion for anaerobic digestion, accuracy in GHG of manure storage and treatment, accuracy in GHG emissions from animal housing. Each method was evaluated and scored and the mean scores obtained for the specific criteria shown in Figure 2(b).The figure shows the highest average score was for Accuracy of manure storage and GHG treatment (2.9 out of 4), followed by Accuracy in GHG emissions due to animal housing (2.3 out of 3) and Leakage inclusion for anaerobic digestion (1.2 out of 3).

The leakage inclusion for anaerobic digestion can only be applied for livestock systems where anaerobic digestion is utilized and indicates whether methane leakage losses have been considered (FAO, 2016b; Grossi et al., 2019). As with the general criteria, the specific criteria also have scoring scales with different maximum scores. For the specific criteria "Leakage inclusion for anaerobic digestion" there was a maximum score of 2 compared to a maximum score of 4 for the other two specific criteria (Table 3).

The methods used for individual GHG emissions (N_2O, CH_4) were then examined, Figure 3 (a) shows that for nitrogen-related emissions, hybrid methods are the most frequent followed by the IPCC Tier 2 method. As for methane, it can be seen from Figure 3(b) that this is also predominantly the case as the most used methods are Hybrid and IPCC Tier 2.

Figure 3 - Methods for estimating emission of N related GHG (a), CH4(b) (Tier 1-3 refer to the IPCC classification (Dong et al., 2006; Gavrilova et al., 2019))



DESCRIPTION OF KEY METHODOLOGIES

Within this section a brief description of each identified LCA methodology is presented. In this research, the different methods are discussed following a tiered approach as proposed by the Intergovernmental panel for the climate change. The FAO LEAP framework proposes three tiers to distinguish the level of complexity: Management (simple empirical) based models (Tier 1); Basic process or complex empirical models (Tier 2); Complex simulation models and direct measurement (Tier 3, FAO, 2020). Direct observations generally belong to Tier 3 methods, while simple emission factors specific to large geographical areas belong to Tier 1, as discussed in previous research (Goglio et al., 2015).

MANURE EMISSIONS (HOUSING AND STORAGE)

The following section presents the methodology used for manure emissions related to housing and storage following these groups in agreement with the FAO LEAP framework (FAO, 2020): i) Management (simple empirical) based models (Tier 1); ii) Basic process or complex empirical models (Tier 2); iii) Complex simulation models and direct measurement (Tier 3). Among the gas considered there is also ammonia, as it was deemed important for manure issues, as a precursor of indirect N2O emissions (Forster et al., 2021; Gavrilova et al., 2019).

Management (simple empirical based models (Tier 1)

IPCC TIER 1 (2006): the IPCC Tier 1 method entails multiplying the total amount of N excretion (from all livestock species/categories) in each type of manure management system by an emission factor for that type of manure management system. Emissions are then summed over all manure management systems. The Tier 1 method is applied using IPCC default N_2O emission factors, default nitrogen excretion data, and default manure management system data (Dong et al., 2006).
The IPCC Tier 1 calculation of N volatilisation in forms of NH_3 and NO_x from manure management systems is based on multiplication of the amount of nitrogen excreted (from all livestock categories) and managed in each manure management system by a fraction of volatilised nitrogen. IPCC Tier 1 annual nitrogen excretion rates should be determined for each livestock category defined by the livestock population characterisation. Country-specific rates may either be taken directly from documents or reports such as agricultural industry and scientific literature or derived from information on animal nitrogen intake and retention (as explained below). In some situations, it may be appropriate to use excretion rates developed by other countries that have livestock with similar characteristics. Finally, for estimating CH_4 emissions from manure management, Tier 1 is a simplified method that only requires livestock population data by animal species/category and climate region or temperature, in combination with IPCC default emission factors, to estimate emissions. Because some emissions from manure management systems are highly temperature dependent, it is good practice to estimate the average annual temperature associated with the locations where manure is managed (Dong et al., 2006).
IPCC Tier 1 methodology (2019): the updated version of the IPCC methodology integrates recent research and development as part of the Global Livestock Environmental Assessment Model (GLEAM) for manure waste management systems, and can applied with country specific parameters developed by the FAO (FAO, 2022; Gavrilova et al., 2019).

For N_2O emissions, the manure management methodology to estimate emissions is based on a larger number of livestock categories, manure management systems and regional areas than within the 2006 IPCC tier 1 methodology (Dong et al., 2006; Gavrilova et al., 2019).

IPCC TIER 2 methodology (2006): the Tier 2 method follows the same calculation equation as Tier 1 but includes the use of country-specific data for some or all of these variables for N_2O emissions from manure management (Dong et al., 2006). For the IPCC Tier 2 methodology, country-specific nitrogen excretion rates for livestock categories are used and it requires more detailed characterisation of the flow of nitrogen throughout the animal housing and manure management systems used in the country. The annual quantity of N excreted by each livestock species/category depends on the total annual N intake and total annual N retention of the animal. Nitrogen intake can also be calculated from feed data and crude protein intake , though default N retention values are provided within a table, as are the fraction of nitrogen in feed taken in by animals which is retained by the different animal species/categories (Dong et al., 2006).

For estimating CH_4 emissions, the IPCC Tier 2 methodology (2006) is a more complex method and requires detailed information on animal characteristics and manure management practices, which is used to develop emission factors specific to the conditions of the country (Dong et al., 2006).

IPCC Tier 2 methodology (2019): this methodology is similar to the IPCC Tier 1 methodology (2019), except that country specific emission factors have been utilised within the equations (Gavrilova et al., 2019).

EMEP/EEA methodology: the EMEP/EEA Air Pollutant Emissions methodology has been developed to calculate emissions from all phases of manure management, including emissions from animal enclosures, open areas, and manure storage facilities, and also emissions that occur after the effluent is applied to the land and from excrement deposited in fields by grazing animals (Amon et al., 2019, 2021). The methodological guidelines take into consideration the various gases (including ammonia, which is responsible for indirect N_2O emission) and non-methane volatile organic compounds (NMVOC) using the calculations within IPCC Tier 1 and IPCC Tier 2 approaches, alongside other methods (Hergoualc'h et al., 2019).

ISPRA method: this methodology is used for estimating national emissions in Italy from the livestock sector and it is based on several methodological guidelines (Amon et al., 2019; Dong et al., 2006; Taurino et al., 2021). The estimation procedure for NH_3 emissions proceeds in successive subtractions from the quantification of nitrogen excreted annually for each livestock category. This quantity can be divided in two different fluxes, depending on whether animals are kept indoor (housing, storage and manure application) or outdoor (grazing). More in detail is incorporated regarding the housing and/or storage system, with a specific emission factor applied to the total nitrogen excretion (Taurino et al., 2021).

U.S. EPA method for stored manure: this method is based on series of coefficients adapted to the US conditions (EPA, 2010). A calculation was developed to estimate the amount of CH_4 emitted from

Anaerobic Digestion (AD) systems utilizing CH_4 capture and combustion technology. First, AD systems were assumed to produce 90% of the maximum CH_4 producing capacity of the manure. This value is applied for all climate regions in US and AD system types. However, this is a conservative assumption as the actual amount of CH_4 produced by each AD system is very variable and will change based on operational and climate conditions. Moreover, an assumption of 90 % is likely overestimating CH_4 production from some systems and underestimating CH_4 production in others. The total amount of CH_4 produced by AD is calculated only as a means to estimate the emissions from AD; i.e., only the estimated amount of CH_4 actually entering the atmosphere from AD is reported in the inventory. The emissions to the atmosphere from AD are a result of leakage from the system (e.g., from the cover, piping, tank, etc.) and incomplete combustion and are calculated using the collection efficiency (CE) and destruction efficiency (DE) of the AD system. The three primary types of AD systems in the United States are covered lagoons, complete mix and plug-flow systems. The CE of covered lagoon systems was assumed to be 75%, and the CE of complete mix and plug flow AD systems was assumed to be 99%. The CH_4 DE from flaring or burning in an engine was assumed to be 98%; therefore, the amount of CH_4 that would not be flared or combusted was assumed to be 2% (EPA, 2010).

UK NARSES: the National Ammonia Reduction Strategy Evaluation System (NARSES) models the flows of total nitrogen and total ammoniacal N (TAN) through the livestock production and manure management system, with NH_3 losses given at each stage as a proportion of the TAN present within that stage (Misselbrook et al., 2016). This method uses emissions factors to estimate ammonia emissions. NARSES was first used to provide the 2004 inventory estimate for UK agriculture, replacing the previously used UK Agricultural Emissions Inventory model (UKAEI) (Misselbrook et al., 2016).

Basic process or complex empirical models (Tier 2)

INITIATOR (Integrated NITrogen Impact Assessment Tool On a Regional scale) is a simple N balance model based on empirical linear relationships between the different N fluxes. The processes and fluxes treated in agricultural soils in INITIATOR include NH_3, NO_x and N_2O emission from housing and manure storage systems. Uptake, immobilisation/mineralisation, nitrification and denitrification in soil - N leaching to ground water and runoff to surface water. The various N from and in agricultural soils are calculated with a consistent set of simple linear equations. INITIATOR is the core model of the model NitroGenius, which is being used as an interactive discussion platform between different stakeholders for governmental policies related to abatement of nitrogen emissions.

HOLOS is a whole-farm software program developed by Agriculture and Agri-Food Canada, this takes into account emissions produced from animal agriculture operations (fertilizer application, tillage, pesticides, etc., AAFC, 2023). It is designed to model beef, dairy, swine, sheep, poultry systems, and several other types of livestock systems. The model has been developed using IPCC Tier 2 (2006) methodology for Canadian condition (Alemu et al., 2017b; Little et al., 2008).

NH_3 Canadian manure model: this model estimates NH_3 emissions using a monthly time scale in Canadian conditions. It includes estimates of daily emission peaks within critical months (Sheppard et al., 2009). The results will contribute to estimates of haze and atmospheric aerosol production, as well as contributions to other potential impacts such as eutrophication of sensitive ecosystems. The key feature of this model is that emissions are expressed as fraction of TAN rather than an emission per animal unit. At each stage, emissions are based only on the on the remaining TAN from the previous stage. Thus, the ammonia losses are based on a mass balance (Sheppard et al., 2009).

Complex simulation models and direct measurements (Tier 3)

IPCC TIER 3 methodology: some countries for which livestock emissions are particularly important went beyond the Tier 2 method and developed models for country-specific methodologies or use measurement–based approaches to quantify emission factors. The method chosen will depend on data availability and national circumstances (Dong et al., 2006). A Tier 3 method utilizes alternative estimation procedures based on a country-specific methodology. For example, a process-based, mass balance approach which tracks nitrogen throughout the system starting with feed input through final use/disposal could be utilized as a IPCC Tier 3 procedure. IPCC Tier 3 methods should be well documented to clearly describe estimation procedures. To reduce uncertainty of the estimates, an IPCC Tier 3 method could be developed with country-specific emission factors for volatilisation and nitrogen leaching and runoff based on actual measurements. All losses of N through manure management systems (both direct and indirect) need to be excluded from the amount of manure N that is available for application to soils.

IFSM: through a major revision, a beef animal component was added along with a crop farm option (no animals) to form the Integrated Farm System Model (IFSM). Several other components were added to simulate environmental impacts including gas emissions, nitrate leaching, and phosphorus runoff and a life cycle assessment to determine the carbon footprint of production systems. Manure emissions are calculated on the basis of the following parameters: type of manure collection, storage, transport, and application (Rotz et al., 2018). Manure CO_2 emissions have been calculated using simple emissions factors specific for manure storage. Methane emissions due to storage have been calculated through a set of empirical equations considering the volume of volatile solids and the temperature. The N_2O emissions for open storage facilities was estimated using simple equations which consider the exposed surface area of the manure storage (Rotz et al., 2018). If the manure dry matter (DM) content is less than 8%, it is assumed that the crust does not form as when manure is loaded onto the top of the stored slurry, or a covered or enclosed tank is used. Therefore, when any of these three issues happen, it is assumed that no N_2O emissions occur (Rotz et al., 2018). For stacked manure, the IPCC Tier 1 emission factor (2006) is used (Dong et al., 2006; Rotz et al., 2018). Ammonia emission rate from storage is calculated on the basis of exposed surface area, total ammonium nitrogen in the manure (TAN) concentration, temperature, air speed, and surface pH (Rotz et al., 2018).

The impact of housing emissions has been calculated considering different types of bedding including straw, sawdust and bedding types. Housing methane and CO_2 emissions due to residue on the barn floor have been estimated using a regression equation which takes into account the amount of manure on the surface of the barn and the temperature (Rotz et al., 2018). Housing N_2O emissions were estimated using IPCC Tier 2 methodologies with the exception of free-stall and tie-stall stables, where the N_2O emissions are assumed to be negligible (Dong et al., 2006; Rotz et al., 2018). Manure ammonia emissions are based on animal housing density, spent time by the animal in the stable, manure removal rate, temperature, air flow (Rotz et al., 2018).

Discussion

IDENTIFIED KEY METHODOLOGICAL ISSUES

Manure emissions (Housing and Storage)

This review aimed to analyse and provide an overview of the methods currently used to assess emissions caused by manure storage. Articles from Europe generally have a greater accuracy as they tend to specify the type of housing, the type of bedding and consequently the manure produced and the subsequent types of manure storage. These details are very important to make the assessment method more accurate, robust, and consequently be reproducible and applicable given the many variables (leakage, storage and treatment, animal housing) that are taken into consideration (Gavrilova et al., 2019; Owen and Silver, 2015; Philippe and Nicks, 2015). What emerged from this analysis was the heterogeneity of results (standard deviations range from -25% to +15%) highlighting the need for a harmonized and more uniform methodology. Indeed, with the goal of assessing and comparing the sustainability of livestock farms on a European scale (at least, given that climate change is global), maintaining different methods does not seem appropriate. Most of the assessed LCA methods received the lowest score in the general criteria regarding applicability. An attempt has been carried out with several limitation by the European Monitoring Evaluation Programme (EMEP)/European Environment Agency (EEA) in Europe, adopting a Tier 2 methodology using emission factors rather than mechanistic models (Amon et al., 2019), which could offer a higher level of accuracy, as discussed previously for soil C in LCAs of cropping systems (FAO, 2018; Goglio et al., 2015).

The applicability of the methods is affected by data requirements. The demand for a multiplicity of data to achieve good completeness and adequate accuracy of results required a specialist knowledge with skills and expertise regardless of the method chosen, given the many variables present in the livestock sector which must be taken into account, as previously discussed for LCAs of cropping systems with regards to soil C (FAO, 2018; Goglio et al., 2015). Indeed the GHG emissions from the livestock sector derive from a large variety of cropping, livestock management and manure management processes (Rotz, 2018). As it is easy to deduce, inappropriate use of the method and incorrect data acquisition and input can cause a distortion of the results and consequently of the emissions produced by the sector. Therefore having the appropriate expertise is essential to obtain high quality data, as previously discussed for LCA of cropping systems and for soil C assessment (FAO, 2018; Goglio et al., 2015). Some methods such as electrochemical sensors that measure gas concentration to have reliable and accurate data must be combined with other methods/equations. However, other methods, such as chamber techniques, present the problem of reproducibility, given the cost and the need for specialized buildings making them therefore impossible to implement and disseminate widely with the current technology. In addition to reproducibility, with the chamber techniques, it is difficult to separate emissions from enteric fermentation (CH_4) and those from manure due to animal housing (CH_4, N_2O, NH_3, NO, N_2) (Tedeschi et al., 2022).

Regarding manure emissions, in the LCA of livestock systems special care should be taken in avoiding double accounting. Indeed if inventory data are used, often different processes in the inventory can include emission at different stage of the manure handling process (housing/yard/pasture/storage/spreading); thus the LCA practitioner should be very careful in the inventory process selection while carrying out the LCA of livestock system in coherence with system boundary of each inventory process and the overall system boundary of the LCA, line with the ISO standards (ISO 2020a,b).

Finally, further assessment of anaerobic digestion processes are necessary as this animal waste management technology can play an important role to develop European strategies for the circular economy and zero waste (Ingrao et al., 2016).

RESEARCH NEEDS AND FUTURE STUDIES

Manure emissions (Housing and storage)

Most of the papers developing methods for manure emissions in LCA of livestock systems were related to European production, demonstrating the European Union's emphasis and focus on greenhouse gas emissions through agreements such as the European Green Deal that sets out how to make Europe the first climate-neutral continent by 2050, in agreement with the conference of parties agreement (UNFCCC, 2015). Given the importance of GHG emissions by European policies, it is essential that the assessment methods are also accurate, reproducible and applicable. Furthermore, it is important to use the most up-to-date methods (i.e. IPCC Tier 3) even if they are more complex, for greater accuracy and reproducibility (Gavrilova et al., 2019). However this is at the expense of their applicability, as highlighted for soil C in LCAs of agricultural systems and livestock assessment (FAO, 2018; Goglio et al., 2015). The application of the most up-to-date methods in the future should also be extended to agricultural consultants and farmers with a more limited level of expertise also through the use of DDS (Decision Support Systems) and throughs modern user friendly software (Bellon-Maurel et al., 2015) to facilitate the monitoring of progress towards mitigation targets and improved efficiencies. However, the lack of uniform methods can encourage criticism of LCA regarding its suitability for livestock systems, due to the large uncertainties which have been identified previously for other agricultural systems (Baustert and Benetto, 2017; Martinelli et al., 2019).

The existing methods and methodologies to measure and estimate GHG emission from manure vary depending on type of management and composition of the manure, type of storage, animal housing, concentration inside and ventilation rate that all contribute to measurement uncertainties (Gavrilova et al., 2019; Grossi et al., 2019; Tedeschi et al., 2022). While "bottom-up" approaches require individual sources to build empirical or mechanistic modelling, "top-down" approaches use models to estimate an emitter contribution from atmospheric measure at an observation point. Regardless of the method used, measurements and associated errors and uncertainty are critical. Therefore, maintaining the development of different approaches seems to be important, while at the same time reinforcing the comparison and the validation of individual methods in different production scenarios, as discussed by previous research for both methane and soil C (FAO, 2018; Tedeschi et al., 2022). This dual development path must meet the need for calibration development and protocol standardization for existing methods.

LCA METHODOLOGICAL GUIDELINES/RECOMMENDATIONS

From the analysis of current LCA methods, some general recommendations can be made regarding the suitability and application of methods when undertaking a livestock system LCA.

For LCA methods regarding the assessment of manure management emissions, the use of direct observations or a Tier 3 method are the most accurate, but in the absence of the necessary data, IPCC 2019 Tier 2 or the EEA 2019 Tier 2 methods are recommended (Amon et al., 2019; Gavrilova et al., 2019). Many different methods have been and can be utilised, therefore when applying an estimation method, limitations should be highlighted and discussed, especially if multiple methods are applied.

Conclusion

In this research an attempt to harmonize LCA methods for livestock systems was successfully carried out. The identified methods for GHG emissions focused on manure emissions and it was generally observed that a high level of accuracy corresponded a low level of applicability and vice-versa. Thus, the choice of the methodology in relation to the LCA objectives is particularly critical to enable high-quality LCAs.

Following the analysis of the available literature, as general recommendation for all the GHG from livestock systems, the choice of LCA methods should be based on the LCA objectives, data availability and expertise of the LCA practitioner. Whilst complex models have been mostly developed for soil C and soil N_2O emissions, for manure emission estimation more complex emission factor equations have been conceived. Whilst IPCC Tier 1 methodologies haves been employed in most of the assessments analysed here, Tier 2 methods, related to the specifics of the manure and housing systems are preferable for improved accuracy. Independently of the method used, method limitations as well as uncertainty analysis undertaken, which should be discussed in all LCAs livestock systems.

Future development of LCA methodologies is necessary to improve LCAs of livestock systems, including the development of improved emission factors or preferably, basic process models which act as a compromise between applicability and accuracy. This LCA method development must be synchronous with improvements of observation methods and the assessment of different crop-livestock management systems.

References

AAFC, A. and A.-F., 2023.Holos software program. https://agriculture.canada.ca/en/agricultural-production/holos-software-program (accessed 23 may 2023).

Alemu, A.W., Amiro, B.D., Bittman, S., MacDonald, D., Ominski, K.H., 2017a. Greenhouse gas emission of Canadian cow-calf operations: A whole-farm assessment of 295 farms. Agr. Syst. 151, 73–83 doi: 10.1016/j.agsy.2016.11.013.

Alemu, A.W., Janzen, H., Little, S., Hao, X., Thompson, D.J., Baron, V., Iwaasa, A., Beauchemin, K.A., Kröbel, R., 2017b. Assessment of grazing management on farm greenhouse gas intensity of beef production systems in the Canadian Prairies using life cycle assessment. Agricultural Systems 158, 1–13 doi: 10.1016/j.agsy.2017.08.003.

Amon, B., Hutchings, N., Dämmgen, U., Sven Sommer, J.W., Seedorf, J., Hinz, T., Hoek, K., Gyldenkærne, S., Mikkelsen, M.H., Dore, C., Jiménez, B.S., Menzi, H., Dedina, M., Haenel, H.-D., Röseman, C., Groenestein, K., Bittman, S., Hobbs, P., Lekkerkerk, L., Bonazzi, G., Couling, S., Cowell, D., Kroeze, C., Pain, B., Klimont, Z., 2019.3.B Manure management. (EMEP/EEA air pollutant emission inventory Guidebook 2019). EEA European Environmental Agency, Copenhagen, Denmark.

Amon, B., Çinar, G., Anderl, M., Dragoni, F., Kleinberger-Pierer, M., Hörtenhuber, S., 2021. Inventory reporting of livestock emissions: the impact of the IPCC 1996 and 2006 Guidelines. Environ. Res. Lett. 16, 075001 doi: 10.1088/1748-9326/ac0848.

Baustert, P., Benetto, E., 2017. Uncertainty analysis in agent-based modelling and consequential life cycle assessment coupled models: A critical review. J. Clean. Prod. 156, 378–394 doi: 10.1016/j.jclepro.2017.03.193.

Bellon-Maurel, V., Peters, G.M., Clermidy, S., Frizarin, G., Sinfort, C., Ojeda, H., Roux, P., Short, M.D., 2015. Streamlining life cycle inventory data generation in agriculture using traceability data and information and communication technologies – part II: application to viticulture. Journal of Cleaner Production 87, 119–129 doi: 10.1016/j.jclepro.2014.09.095.

Bhatt, A., Abbassi, B., 2021. Review of environmental performance of sheep farming using life cycle assessment. J. Clean. Prod. 293, 126192 doi: 10.1016/j.jclepro.2021.126192.

Bohran, M., Mukhtar, S., Capareda, S., Rahm, S., 2012. Greenhouse Gas Emissions from Housing and Manure Management Systems at Confined Livestock Operations, in: Marmolejo Rebellon, L.F. (Ed.), Waste Management - An Integrated Vision. InTech doi: 10.5772/51175.

Brady, N., Weil, R., 2002.The Nature and Properties of Soils, 13th ed. Prentice Hall, Upper Saddle River, New Jersey, USA.

Bui, M., Adjiman, C.S., Bardow, A., Anthony, E.J., Boston, A., Brown, S., Fennell, P.S., Fuss, S., Galindo, A., Hackett, L.A., Hallett, J.P., Herzog, H.J., Jackson, G., Kemper, J., Krevor, S., Maitland, G.C., Matuszewski, M., Metcalfe, I.S., Petit, C., Puxty, G., Reimer, J., Reiner, D.M., Rubin, E.S., Scott, S.A., Shah, N., Smit, B., Trusler, J.P.M., Webley, P., Wilcox, J., Mac Dowell, N., 2018. Carbon capture and storage (CCS): the way forward. Energy Environ. Sci. 11, 1062–1176 doi: 10.1039/C7EE02342A.

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. Anim. 7, 212–219 doi: 10.1017/S1751731113000785.

Don, A., Seidel, F., Leifeld, J., Kätterer, T., Martin, M., Pellerin, S., Emde, D., Seitz, D., Chenu, C., 2024. Carbon sequestration in soils and climate change mitigation—Definitions and pitfalls. Global Change Biology 30, e16983 doi: 10.1111/gcb.16983.

Dong, H., Mangino, J., McAllister, T.A., Hatfield, J.L., Johnson, D.E., Lassey, K.R., Lima, M.A., Romanovskaya, A., Bartram, D., Gibb, D., Martin, J.H., Jr, 2006.Chapter 10: Emissions from Livestock and Manure Management, IPCC guidelines for national inventories. Intergovernmental Panel for Climate Change (IPCC), Geneva.

EPA, 2010.Report on 2010 U.S. Environmental Protection Agency (EPA) Decontamination Research and Development Conferenc, EPA/600/R-11/052. U.S. Environmental Protection Agency, Washington, DC, USA.

FAO, 2016a.Environmental Performance of Pig Supply Chains: Guidelines for assessment (Livestock Environmental 251 Assessment and Performance Partnership). Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016b.Environmental performance of animal feeds supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016c.Greenhouse gas emissions and fossil energy use from small ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016d.Greenhouse gas emissions and fossil energy use from poultry supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016e.Environmental performance of large ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2017.Livestock solutions for climate change. Food and Agriculture Organisation of the United Nations, Rome, Italy.

FAO, 2018.Measuring and modelling soil carbon stocks and stock changes in livestock production systems – Guidelines for assessment (Draft for public review). Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2020.Livestock Environmental Assessment and Performance (LEAP) Partnership | Food and Agriculture Organization of the United Nations. http://www.fao.org/partnerships/leap/en/ (accessed 11 may 2020).

FAO, 2022.Global Livestock Environmental Assessment Model (GLEAM). https://www.fao.org/gleam/en/ (accessed 9 august 2022).

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2012. The interaction between milk and beef production and emissions from land use change – critical considerations in life cycle assessment and carbon footprint studies of milk. J. Clean. Prod. 28, 134–142 doi: 10.1016/j.jclepro.2011.11.046.

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M., Zhang, H., 2021. The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity, in: Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S.L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J.B.R., Maycock, T.K., Waterfield, T., Yelekçi, O., Yu, R., Zhou, B. (Eds.), Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 923-1054, doi: 10.1017/9781009157896.009.

Gavrilova, O., Leip, A., Dong, H., MacDonald, J., Gomez Bravo, C., Amon, B., Barahona Rosales, R., Del Prado, A., De Lima, M., Oyhantçabal, W., Van Der Weerden, T., Widiawati, Y., Bannink, A., Beauchemin, K., Clark, H., Leytem, A., Kebreadìb, E., Ngwabie, N., Imede Opio, C., Vanderzaag, A., Vellinga, T., 2019.2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Chapter 10: Emission from livestock and manure management. IPCC, Interngovernmental Panel for Climate Change, Geneva.

Godfray, H.C.J., Aveyard, P., Garnett, T., Hall, J.W., Key, T.J., Lorimer, J., Pierrehumbert, R.T., Scarborough, P., Springmann, M., Jebb, S.A., 2018. Meat consumption, health, and the environment. Sci. 361, eaam5324 doi: 10.1126/science.aam5324.

Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J. Clean. Prod. 104, 23–39 doi: 10.1016/j.jclepro.2015.05.040.

Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2017. Addressing crop interactions within cropping systems in LCA. Int. J. Life Cycle Assess. 1–9 doi: 10.1007/s11367-017-1393-9.

Goglio, P., Knudsen Trydeman, M., 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. Letters 4, 100035 doi: 10.1016/j.clpl.2023.100035.

Google, 2024.Google Forms – create and analyse surveys, for free. https://www.google.com/intl/en-GB/forms/about/ (accessed 13 january 2022).

Grossi, G., Goglio, P., Vitali, A., Williams, A.G., 2019. Livestock and climate change: impact of livestock on climate and mitigation strategies. Anim. Frontiers 9, 69–76 doi: 10.1093/af/vfy034.

Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., Del Prado, N., Kasimir, A., MacDonald, D., Ogle, S., Regina, K., van der Weerden, T., Liang, C., Noble, A., 2019. N2O emissions from managed soils, and CO2 emissions from lime and urea application., in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Interngovernmental Panel for Climate Change, Geneva.

Ingrao, C., Bacenetti, J., Bezama, A., Blok, V., Geldermann, J., Goglio, P., Koukios, E.G., Lindner, M., Nemecek, T., Siracusa, V., Zabaniotou, A., Huisingh, D., 2016. Agricultural and forest biomass for food, materials and energy: bio-economy as the cornerstone to cleaner production and more sustainable consumption patterns for accelerating the transition towards equitable, sustainable, post fossil-carbon societies. Journal of Cleaner Production 117, 4–6 doi: 10.1016/j.jclepro.2015.12.066.

IPCC, 2022. Climate change 2022: Mitigation of climate change. WGIII Mitigation of Climate Change Climate Change 2022 Working Group III contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, Interngovernmental panel for climate change, Geneva, Switzerland.

ISO, 2020a. EN ISO 14044:2006+A2:2020. Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organization for Standardization, Geneva.

ISO, 2020b. EN ISO 14040:2006+A1 Environmental Management- Life Cycle Assessment, Principles and Framework. International Organization for Standardization, Geneva.

Jourdaine, M., Loubet, P., Trebucq, S., Sonnemann, G., 2020. A detailed quantitative comparison of the life cycle assessment of bottled wines using an original harmonization procedure. J. Clean. Prod. 250, 119472 doi: 10.1016/j.jclepro.2019.119472.

Kalhor, T., Rajabipour, A., Akram, A., Sharifi, M., 2016. Environmental impact assessment of chicken meat production using life cycle assessment. Inf. Process. Agric. 3, 262–271 doi: 10.1016/j.inpa.2016.10.002.

Kim, D., Thoma, G., Nutter, D., Milani, F., Ulrich, R., Norris, G., 2013. Life cycle assessment of cheese and whey production in the USA. Int J Life Cycle Assess 18, 1019–1035 doi: 10.1007/s11367-013-0553-9.

Kristensen, T., Søegaard, K., Eriksen, J., Mogensen, L., 2015. Carbon footprint of cheese produced on milk from Holstein and Jersey cows fed hay differing in herb content. J. Clean. Prod. 101, 229–237 doi: 10.1016/j.jclepro.2015.03.087.

Lal, R., Stewart, B.A. (Eds.), 2018. Soil and Climate, 1st ed. CRC Press, Boca Raton. doi: 10.1201/b21225

Little, S.M., Lindeman, J., Maclean, K., Janzen, H.H., 2008.Holos - A tool to estimate and reduce GHGs from farms. Methodology and algorythms for version 1.1. AAFC Agriculture and Agri-Food Canada, Ottawa.

López-Andrés, J.J., Aguilar-Lasserre, A.A., Morales-Mendoza, L.F., Azzaro-Pantel, C., Pérez-Gallardo, J.R., Rico-Contreras, J.O., 2018. Environmental impact assessment of chicken meat production via an integrated methodology based on LCA, simulation and genetic algorithms. J. Clean. Prod. 174, 477–491 doi: 10.1016/j.jclepro.2017.10.307.

Macombe, C., Loeillet, D., Gillet, C., 2018. Extended community of peers and robustness of social LCA. Int. J. Life Cycle Assess. 23, 492–506 doi: 10.1007/s11367-016-1226-2.

Martinelli, G. do C., Schlindwein, M.M., Padovan, M.P., Vogel, E., Ruviaro, C.F., 2019. Environmental performance of agroforestry systems in the Cerrado biome, Brazil. World Dev. 122, 339–348 doi: 10.1016/j.worlddev.2019.06.003.

McAuliffe, G.A., Chapman, D.V., Sage, C.L., 2016. A thematic review of life cycle assessment (LCA) applied to pig production. Environ. Impact Assess. Rev. 56, 12–22 doi: 10.1016/j.eiar.2015.08.008.

McConkey, B., Ogle, S.M., Chirinda, N., Kishimoto-Mo, A.W., Baldock, J., Trunov, A., Alsaker, C., Lehmann, J., Woolf, D. (Eds,), 2019. Chapter 6: Grassland, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC, Intergovernmental panel for climate change, Geneva, Switzerland.

Misselbrook, T.H., Gilhespy, S.L., Cardenas, L.M., Williams, J., Dragosits, U., 2016.Inventory of Ammonia Emissions from UK Agriculture, DEFRA Contract SCF0102, Inventory submission report. Department of Environment, Food and Rural Affairs, London, UK.

Morris, J., Brown, S., Cotton, M., Matthews, H.S., 2017. Life-Cycle Assessment Harmonization and Soil Science Ranking Results on Food-Waste Management Methods. Environ. Sci. Technol. 51, 5360–5367 doi: 10.1021/acs.est.6b06115.

Oertel, C., Matschullat, J., Zurba, K., Zimmermann, F., Erasmi, S., 2016. Greenhouse gas emissions from soils—A review. Geochemistry 76, 327–352 doi: 10.1016/j.chemer.2016.04.002.

Ogle, S., Wakelin, S.J., Buendia, L., McConckey, B., Baldock, J., Akiyama, H., Kishimoto-Mo, A.M., Chirinda, N., Bernoux, M., Bhattacharya, S., Chuersuwan, N., Goheer, M.A.R., Hergoualc'h, K., Ishizuka, S., Lasco, R.D., Pan, X., Pathak, H., Regina, K., Sato, A., Vazquez-Amabile, G., Wang, C., Zheng, X., Alsaker, C., Cardinael, R., Corre, M.D., Gurung, R., Mori, A., Lehmann, J., Rossi, S., Van Straaten, O., Veldkamp, E., Woolf, d., Yagi, K., Yan, X., 2019a. Chapter 5: Cropland, in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC International Panel on Climate Change, Geneva.

Ogle, S., Kurz, W.A., Green, C., Brandon, A., Baldock, J., Domke, J., Herold, M., Bernoux, M., Chirinda, N., De Ligt, R., Federici, S., Garcia, E., Grassi, G., Gschwantner, T., Hirata, Y., Houghton, R., House, J.J., Ishizuka, S., Jonckheere, I., Krisnawati, H., Lehtonen, A., Kinyanjui, M.J., McConkey, B., Naesset, E., Niinistö, S.M., Ometto, J.P., Panichelli, L., Paul, T., Peterson, H., Reddy, S., Regina, K., Rocha, M., Rock, J., Sanz-Sanchez, M., Sanquetta, S., Sato, S., Somogyi, Z., Trunov, A., Vazquez-Amabile, G., Vitullo, M., Wang, C., Waterworth, R.M., Collet, M., Harmon, M., Lehmann, J., Shaw, C.H., Shirato, Y., Wolf, D., 2019b. Chapter 2: Generic methodologies, applicable to multiple land-use categories., in: 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories,. IPCC, Intergovernmental panel for climate change, Geneva.

Owen, J.J., Silver, W.L., 2015. Greenhouse gas emissions from dairy manure management: a review of field-based studies. Glob. Chang. Biol. 21, 550–565 doi: 10.1111/gcb.12687.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P., Smith, P., 2016. Climate-smart soils. Nature 532, 49–57 doi: 10.1038/nature17174.

Peters, G.M., Rowley, H.V., Wiedemann, S., Tucker, R., Short, M.D., Schulz, M., 2010. Red Meat Production in Australia: Life Cycle Assessment and Comparison with Overseas Studies. Environ. Sci. Technol. 44, 1327–1332 doi: 10.1021/es901131e.

Petersen, B.M., Knudsen, M.T., Hermansen, J.E., Halberg, N., 2013. An approach to include soil carbon changes in life cycle assessments. J. Clean. Prod. 52, 217–224 doi: 10.1016/j.jclepro.2013.03.007.

Philippe, F.-X., Nicks, B., 2015. Review on greenhouse gas emissions from pig houses: Production of carbon dioxide, methane and nitrous oxide by animals and manure. Agr. Ecosys. Environ. 199, 10–25 doi: 10.1016/j.agee.2014.08.015.

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Sci. 360, 987–992 doi: 10.1126/science.aaq0216.

Rotz, C.A., 2018. Modeling greenhouse gas emissions from dairy farms. J. Dairy Sci. 101, 6675–6690 doi: 10.3168/jds.2017-13272.

Rotz, C.A., Corson, M.S., Chianese, D.S., Montes, F., Hafner, S.D., Bonifacio, H.F., Coiner, C.U., 2018.Integrated Farm System Model: Reference Manual. USDA Agricultural Research Serv., University Park, PA, US.

Segura-Salazar, J., Lima, F.M., Tavares, L.M., 2019. Life Cycle Assessment in the minerals industry: Current practice, harmonization efforts, and potential improvement through the integration with process simulation. J. Clean. Prod. 232, 174–192 doi: 10.1016/j.jclepro.2019.05.318.

Sheppard, S.C., Bittman, S., Tait, J., 2009. Monthly NH 3 emissions from poultry in 12 Ecoregions of Canada. Can. J. Anim. Sci. 89, 21–35 doi: 10.4141/CJAS08055.

Siegert, M.-W., Lehmann, A., Emara, Y., Finkbeiner, M., 2019. Harmonized rules for future LCAs on pharmaceutical products and processes. Int. J. Life Cycle Assess. 24, 1040–1057 doi: 10.1007/s11367-018-1549-2.

Skunca, D., Tomasevic, I., Nastasijevic, I., Tomovic, V., Djekic, I., 2018a. Life cycle assessment of the chicken meat chain. Journal of Cleaner Production 184, 440–450 doi: 10.1016/j.jclepro.2018.02.274.

Skunca, D., Tomasevic, I., Nastasijevic, I., Tomovic, V., Djekic, I., 2018b. Life cycle assessment of the chicken meat chain. J. Clean. Prod. 184, 440–450 doi: 10.1016/j.jclepro.2018.02.274.

Sykes, A.J., Macleod, M., Eory, V., Rees, R.M., Payen, F., Myrgiotis, V., Williams, M., Sohi, S., Hillier, J., Moran, D., Manning, D.A.C., Goglio, P., Seghetta, M., Williams, A., Harris, J., Dondini, M., Walton, J., House, J., Smith, P., 2019. Characterising the biophysical, economic and social impacts of soil carbon sequestration as a greenhouse gas removal technology. Glob. Change Biol. 0 doi: 10.1111/gcb.14844.

Taki, R., Wagner-Riddle, C., Parkin, G., Gordon, R., VanderZaag, A., 2019. Comparison of two gap-filling techniques for nitrous oxide fluxes from agricultural soil. Can. J. Soil. Sci. 99, 12–24 doi: 10.1139/cjss-2018-0041.

Taurino, E., Bernetti, A., Caputo, A., Cordella, M., Lauretis, R., D'Elia, I., Cristofaro, E., Gagna, A., Gonella, B., Moricci, F., Peschi, E., Romano, D., Vitullo, M., 2021.Italian emission inventory 1990-2019. Informative inventory report 2021. ISPRA report 342/2021, Institute for Environmental Protection and Research (ISPRA), Rome, Italy.

Tedeschi, L.O., Abdalla, A.L., Álvarez, C., Anuga, S.W., Arango, J., Beauchemin, K.A., Becquet, P., Berndt, A., Burns, R., De Camillis, C., Chará, J., Echazarreta, J.M., Hassouna, M., Kenny, D., Mathot, M., Mauricio, R.M., McClelland, S.C., Niu, M., Onyango, A.A., Parajuli, R., Pereira, L.G.R., del Prado, A., Paz Tieri, M., Uwizeye, A., Kebreab, E., 2022. Quantification of methane emitted by ruminants: a review of methods. Journal of Animal Science 100, skac197 doi: 10.1093/jas/skac197.

UNFCCC, 2015.Conference of Parties Agreement, 21st session, Paris Agreement. United Nation Framework Convention on Climate Change, Paris.

Vagnoni, E., Franca, A., Breedveld, L., Porqueddu, C., Ferrara, R., Duce, P., 2015. Environmental performances of Sardinian dairy sheep production systems at different input levels. Sci. Tot. Environ. 502, 354–361 doi: 10.1016/j.scitotenv.2014.09.020.

Williams, A.G., Leinonen, I., Kyriazakis, I., 2016. Environmental benefits of using turkey litter as a fuel instead of a fertiliser. J. Clean Prod. 113, 167–175 doi: 10.1016/j.jclepro.2015.11.044.

Zampori, L., Pant, R., 2019.Suggestions for updating the Product Environmental Footprint (PEF) method. EUR 29682 EN, Publications Office of the European Union, Luxembourg.

SG5d: Evaluating Life Cycle Assessment methodologies for enteric methane emissions

Manuel Romero-Huelva[j], Simon Moakes[b,c], Mariano Pauselli[a], Emanuele Lilli[a], Pietro Goglio[a], Laurence G. Smith[g, k], David Yanez-Ruiz[j]

[a]Department of Agricultural, Food, and Environmental Sciences, University of Perugia, Borgo XX Giugno 74, 06121 Perugia (PG), Italy
[b]Department of Food System Sciences, Research Institute of Organic Agriculture (FiBL), Frick, Switzerland
[c]IBERS, Aberystwyth University, UK
[g]School of Agriculture, Policy and Development, University of Reading, UK
[j]Estación Experimental del Zaidín (CSIC), Profesor Albareda 1, 18008 Granada, Spain
[k]Department of Biosystems and Technology, Swedish University of Agricultural Sciences, Box 190, SE-234 22 Lomma, Sweden

Abstract

Worldwide greenhouse gas emissions (GHG) reached 59 Gt of CO_2 eq in 2019 with an increasing demand for livestock sector GHG emissions to be reduced. LCA has been successful in assessing GHGs from livestock systems. However, no harmonization attempt has been carried out, despite the need to improve LCA methodologies for assessing GHG in the LCA of livestock systems. LCA methods in livestock systems and production chains include the quantification of one of the main sources of GHG emissions, methane, arising from enteric fermentation within the digestive systems of livestock.

We therefore undertook a review of existing methods to develop a coherent harmonisation approach. The approach adopted in this study was based on two anonymous expert surveys and a series of expert workshops (n=21) to define general and specific criteria to review LCA methods for GHG emissions used in LCA of livestock systems. More than 3,232 scientific papers and reports were identified, 118 were screened and 36 included in the final review.

The results showed that a high level of accuracy corresponded to a low level of applicability and vice versa. Thus, the choice of the methodology in relation to the LCA objectives is a particularly critical for a high quality LCA assessment. Following the analysis of the available literature, a general recommendation for all the GHG from livestock systems, the choice of LCA methods should be based on the LCA objectives, data availability and expertise of the LCA practitioner. Enteric emissions would ideally be directly assessed with equipment, but in the absence of suitable facilities,

at minimum of a Tier 2 method should be adopted to estimate enteric emissions, based on animal and diet parameters. Further developments to allow more use of in-situ methane measurements would improve the accuracy of LCAs for specific livestock management strategies.

Introduction

Worldwide greenhouse gas emissions reached 59 Gt of CO_2eq in 2019 with agriculture forestry and land use sector contributing around 22% of total emissions. Thus, there is an increasing demand for greenhouse emission reduction for every sector of the economy, including agriculture (IPCC, 2022). At the same time, worldwide demand for animal products is predicted to double over the next decades due to population growth and increasing economic prosperity (Godfray et al., 2018).

It is estimated that livestock supply chains are responsible for 14.5% of all anthropogenic greenhouse gas emissions (FAO, 2017). Within the sector, feed production, manure management and enteric fermentation are the main contributors to climate change impacts. N_2O and CH_4 emissions from manure management contributed 4.3% and 5.7% to global greenhouse gas emissions of livestock production chains respectively, while CH_4 from enteric fermentation for 44.1% of the total livestock emissions (FAO, 2017). N_2O emissions from the application and deposition of manure and nitrous oxide emissions from fertilizers and crop residues in feed production contributed for 13.4% and 5.8% to the livestock sector's emissions respectively, while CO_2 emissions from feed production contributed 13% (FAO, 2017). In addition to the GHG emissions, soil contains the largest share of terrestrial carbon under a dynamic equilibrium which depends *inter alia* on soil types, climate, and management practices.

Methane is the main greenhouse-gas contributor to global warming in the livestock sector; it is generated in the digestive tract of livestock during the microbial fermentation of feed components. Anaerobic fermentation in the different sections of the gut, and the methane concentration differ significantly among animal species (de la Fuente et al., 2019), though ruminants generate most of it, due to processes within the rumen as carbohydrates are fermented. Methane is produced by certain types of microorganisms (methanogens), and the microbial species composition is largely affected by diet, location, host and the gut section. The process uses hydrogen and carbon dioxide from carbohydrates fermentation to produce methane. The amount of methane released depends on mostly on the type and quantity of the feed consumed but also on the age and weight of the animal, and (Gavrilova et al., 2019).

Life cycle assessment (LCA) is globally recognized as the leading method to measure the environmental impacts of products, processes, or services, as it can quantify a wide range of themes and provide a deep understanding of impacts, from cradle to grave. LCA is an assessment method commonly used to assess livestock systems and products due to its ability to identify environmental hotspot and trade-offs across different types of pollution (Cederberg et al., 2013). LCA has been also widely used to assess climate change impacts of food and livestock products (Grossi et al., 2019; Poore and Nemecek, 2018).

Several harmonisation attempts were carried out in sectors other than agriculture (Segura-Salazar et al., 2019; Siegert et al., 2019), while others focused on wines (Jourdaine et al., 2020) or food waste advocating for a better integration between LCA and soil science (Morris et al., 2017). No harmonization attempt has been made for soil C, soil N_2O emissions, manure emissions and enteric fermentation. Although, recent guidelines have been proposed by the Food and Agriculture Organisation (FAO, 2016a, 2016b, 2016c, 2016d, 2016e, 2020). However, these reports are mostly prescriptive (i.e.: suggesting methodology) and they contain a limited comparison and discussion of methods.

Within this study, we undertook a coherent review process for GHGs to assess LCA methods in livestock systems focused on enteric fermentation. The overall aim of this work was to present advantages or disadvantages of the LCA methods reviewed to inform LCA practitioners and researchers to identify future methodological research needs.

Methodology

SEARCH, SCREENING CRITERIA AND DATA PROCESSING

Search criteria

A literature search was conducted using Scopus, Elsevier, Google Scholar and Web of science search engines. The search terms and search term combinations employed were described below in table 1 and included all papers published between 2012 to 2022.

Screening and review procedures

The collected sources were screened against the following criteria:

- Peer-reviewed publication in a scientific journal or published by FAO, European Commission
- English language publication
- Method is related to and applicable for LCA
- Method is related to livestock systems or its components
- Method is applicable for European livestock systems

A systematic review of the existing literature, based on the methodology described above was conducted to provide a comprehensive assessment on how LCA methodologies include the issue of livestock GHG emissions related to enteric fermentation. This critical review sought to identify the most significant components and summarize the main concepts. To achieve this, a review protocol was developed (Figure 1), describing the search and screening process including an iterative process of article selection based on restrictive criteria.

In the first stage (i.e. "*identification step*"), the literature search was performed, according to the queries defined in Table 1, in Scopus, Web of Science and Google Scholar databases. Searches led to a total of 3,232 papers. Only articles published during the 2012-2022 period in the following research areas: agriculture or animals or cattle or dairying or animal feed or animal husbandry or swine or livestock or chickens or poultry. When the Google search engine was used, the selection of papers was stopped at page 15 of the search results. Papers with research not fully relevant for the livestock sector such as rice, plastic, biofuel, bioenergy were excluded. Energy papers related to biogas, insects, fish, or feed production without any focus on livestock were also excluded.

Table 1: Combinations of search terms for the subgroup "GHG Emission Issues"

Database	Combination	Search strings[1]
Scopus & Web of Science	1	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("enteric fermentation")
	2	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("soil*") AND ("emissions" OR "nitrous oxide" OR "N2O" OR "carbon dioxide" OR "CO2" OR "carbon sequestration" OR "GHG" OR "greenhouse gas*" OR "C dynamics" OR "soil) AND ("carbon") AND ("livestock")
	3	("Life Cycle Assessment" OR " life cycle analysis") AND ("livestock" OR "dairy" OR "cattle" OR "sheep" OR "pig*" OR "poultry" OR "goat*" OR "milk" OR "egg*" OR "chicken*" OR "cow*" OR "husbandry") AND ("emissions") NOT ("waste" OR "biofuel" OR "bioenergy")
	4	("LCA" OR "Life Cycle Assessment" OR " life cycle analysis") AND ("emissions") AND ("livestock*" OR "dairy" OR "sheep" OR "pig" OR "poultry" OR "goat" OR "milk" OR "egg*" OR "Chicken" OR "cow "NOT "waste" OR "biofuel" OR "bioenergy")
Google Scholar	5	"LCA" "enteric fermentation" OR "enteric emissions"
	6	"LCA" "livestock"

[1]Last accessed in March 2022

Reviewing abstracts and titles, access, language, region and then removing duplicate papers. The second step of "Screening" was made by using restrictive criteria ("refine results") excluding appearances before 2012 and papers without access (1,175 papers), and a second selection was performed based on the content of the abstract, excluding off-topic. Finally, 621 papers were selected as "Eligible" for a full-text reading. The last step, following the full-text reading, excluded papers that did not directly use in the LCA application or did not focus on the key topic of "GHG emissions". Through this iterative process, the total amount of papers included in the qualitative analysis was reduced to 36 papers.

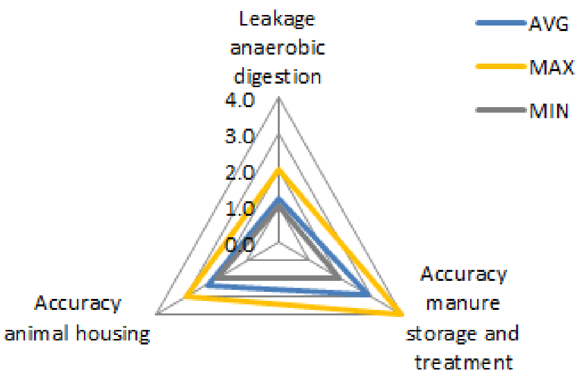


Figure 1: Methodological steps of the literature search process

GENERAL CRITERIA SELECTION

The papers identified as part of this harmonization were then reviewed using both general criteria and specific criteria to assess the LCA methods for livestock systems and product. General criteria used in the harmonization of LCA methods for livestock systems for GHG emissions were selected using a participatory approach based on a modified DELPHI method (Goglio et al., 2023). The selection of key topics was carried out through an anonymous survey which allowed us to screen the various topics and provide a priority list on the basis of a preliminary literature review with key words: "life cycle assessment", "livestock", "poultry", "beef", "dairy", "milk", "cheese", "meat", "pig", "pork", "turkey", "sheep", "lamb" and "goat", "methods", "harmonisation", "review", "methodology". Within the survey each participant was invited to express a priority value with a range from one (low priority) to ten (high priority). On this basis, specific criteria and review approaches were developed for each key topic.

A review of frameworks used to assess LCA methods was undertaken, and key search words included: "LCA methods", "LCA framework", "livestock", "agriculture". Articles and publications were collected from literature including the FAO LEAP reports and the PEFCR general guidelines (FAO, 2018; Zampori and Pant, 2019). Only publicly available documents were screened.

An anonymous survey was carried out using Google survey (Google, 2024), involving LCA experts. The general criteria selected through the survey were then further screened through LCA expert discussions to ensure that both the definition and the scale would be coherent with the harmonization efforts of the LCA methodology for livestock systems and products. For some criteria the definition and the scale was reformulated and modified to ensure rigour and coherence in the review of the LCA methods (Goglio et al., 2023). Table 2 presents the set of general criteria defined for the harmonization of LCA methods for livestock systems.

Table 2: Matrix of general criteria description and the correspondent scale used for the critical review of LCA methodologies

General criteria and definition	Level 1	Level 2	Level 3	Level 4
Transparency and reproducibility: Comprehensive documentation and mechanisms that allow reviewers to verify/review all data, calculations, and assumptions	LCA methodologies which do not allow reviewers to verify/review the results, calculations, and assumptions.	LCA methodologies which could be reviewed together with the results, but some calculations and assumptions cannot be reviewed.	LCA methodologies which fully allows reviewers to verify/review the results, calculations, and assumptions	
Completeness: Relationships between quantification of the environmental impact (material/energy flows and other environmental interventions) and adherence to the defined system boundary, the data requirements, and the impact assessment methods employed	The quantification of the environmental impacts including all material/energy flows and other environmental interventions do not have adherence to the system boundary, the data requirements and the impact assessment methods employed	The quantification of the environmental impacts is conformed either to the defined system boundary or the data requirements or the system method employed;	The quantification of the environmental impacts conforms to two aspects between the defined system boundary, data requirements and impact assessment method employed.	The quantification of the environmental impacts fully corresponds to the system boundary, data requirements and the impact assessment methods employed.
Fairness and acceptance: Level playing field across competing products, processes, and industries	The LCA methodology does not provide level playing field across products, processes, and industries;	The LCA methodology provides a level playing field for at least two products, processes, and industries (e.g., Beef and diary, beef and pig);	LCA provides a level playing field for several products, processes, and industries.	
Robustness: Associated in the RACER framework the following sub criteria of providing a defensible theory, Sensitivity, Data quality, Reliability, Consistency, Comparability, Boundaries	The LCA methodology is not based on defensible theory, lacks sensitivity on certain environmental impacts either because of its reliability, comparability, the chosen system boundary, or its comparability.	The LCA methodology is based on a defensible theory, but it lacks sensitivity, reliability, comparability and it is not in agreement with the system boundaries.	The LCA methodology is based on a defensible theory with a satisfactory sensitivity, reliability, data quality, consistency, comparability and in agreement with the system boundaries.	

Applicability: Ability of the method to be used by a wide range of LCA practitioners	The LCA method can only be used with LCA expertise and extensive data availability	The LCA method can be used with either limited LCA expertise or data availability	The LCA method can be easily used with very limited LCA expertise and data availability

SPECIFIC CRITERIA IDENTIFICATION

Following the definition of the general criteria, specific evaluation criteria were defined for each specific topic in four workshops. The definition and the scale of specific criteria were reformulated and modified to ensure rigour and coherence in the review of the LCA methods. The expert discussions were conducted as a community of peers (Macombe et al., 2018) and different specific criteria were selected for CH_4 emissions originating from enteric fermentation.

To undertake this, a systematic literature review was conducted to assess how methane emissions from enteric fermentation are evaluated within the LCA of livestock systems.

Enteric fermentation emissions are generated as a result of microbial fermentation of the feed in the digestive tract. In particular, the amount of methane released depends on many aspects, both related to the livestock species and dietary composition (Gavrilova et al., 2019). A specific criteria was formulated to address how LCA methods assess enteric fermentation in the LCA of livestock systems. The specific criteria selected for *"Enteric fermentation"* is reported in Table 3.

Table 3: Matrix of specific criteria description and the correspondent scale used to assess consideration of enteric emissions in LCA methodologies.

Description of the specific criterion	**Level 1**	**Level 2**	**Level 3**
Accuracy: It is the level of details that the LCA methods can capture in the assessment of GHG emissions.	The LCA method can be approximated with a Tier 1 IPCC (2019) level (Gavrilova et al., 2019), which is based on a simplified approach which relies on default emission factors and a basic characterization of the livestock population (e.g. animal species, number of animals, possible definition of low/high productivity system);	The LCA method can be approximated with a Tier 2 IPCC (2019) level, therefore distinguishing livestock population into subcategories for each species according to age, type of production, and sex (e.g., adult dairy, other adult, and growing cattle; growing / fattening animal stage; indoor facilities vs. grown under grazing conditions). Level 2 requires feed intakes for a representative animal in each subcategory (which considers the digestibility of the feed ingredients, the energy intake and use, …). The emission factors for each category of livestock are estimated based on the gross energy intake and methane conversion factor for the category (Ym).	The LCA method can be approximated with a Tier 3 IPCC (2019) level which is based on methods which consider additional factors affecting feed requirements and/or consumption to level 2 (e.g. heat and cold stress, animal breed types, variations in feed digestibility and chemical composition, factors affecting the digestive system (and microbiome) of the animal).

Results

QUANTITATIVE RESULTS

Enteric fermentation emissions are considered the main sources of methane and mainly involve ruminant livestock due to their specialised digestive physiology, while the emissions from poultry and pigs are negligible by comparison. The total number of papers considered in the study was 36, with 58% of them related to dairy cattle, 19% beef cattle, whilst 6% involved sheep, pigs, combined dairy and beef and generalist papers on the LCA application to livestock production systems respectively, and only 3% for goat farming. Five general criteria (Transparency and Reproducibility, Completeness, Fairness and Acceptance, Applicability) and one Specific (Accuracy) criteria were considered to evaluate the papers and methods used to estimate enteric emissions, respectively. The results highlighted that whilst all the general criteria measured were near to the maximum level for each method, the rating for accuracy was at a lower level (average 2.11 on 3 level scale) (Figure 2).

Within LCA assessments in dairy cattle, the evaluation of the 20 reviewed papers found seven different types of focus. These included: 1) survey research (five papers and one of them also considered beef), 2) comparison among different rearing systems or different farm strategies (two papers), 3) case studies (two papers), 4) impact of different physiological status on GHG emissions (two papers), 5) methodological papers (one paper), 6) comparison among different simulated scenarios (six papers), 7) comparison between different feeding strategies (two papers). The geographic regions involved were mainly Europe, Northern America, and Australia. There were 11 methods used to estimate the enteric methane emissions and among them the most used was IPCC 2006 Tier 2 (36.36%), the more detailed version (Tier 2b: 4.05%) or its adjustment based on local (9.09%) raising and/or feeding systems with some modification of the Ym. value. Other methods were based on empirical equations derived from specific experiments aimed at directly estimating the enteric methane emissions (Figure 2) or daily methane emissions estimated according to equations based on the ingested diet characteristics as proposed by Mills et al., (2003), Ellis et al., (2007) and Yan et al., (2000, 2006), or directly estimated in respiratory chambers (Grandl et al., 2019).

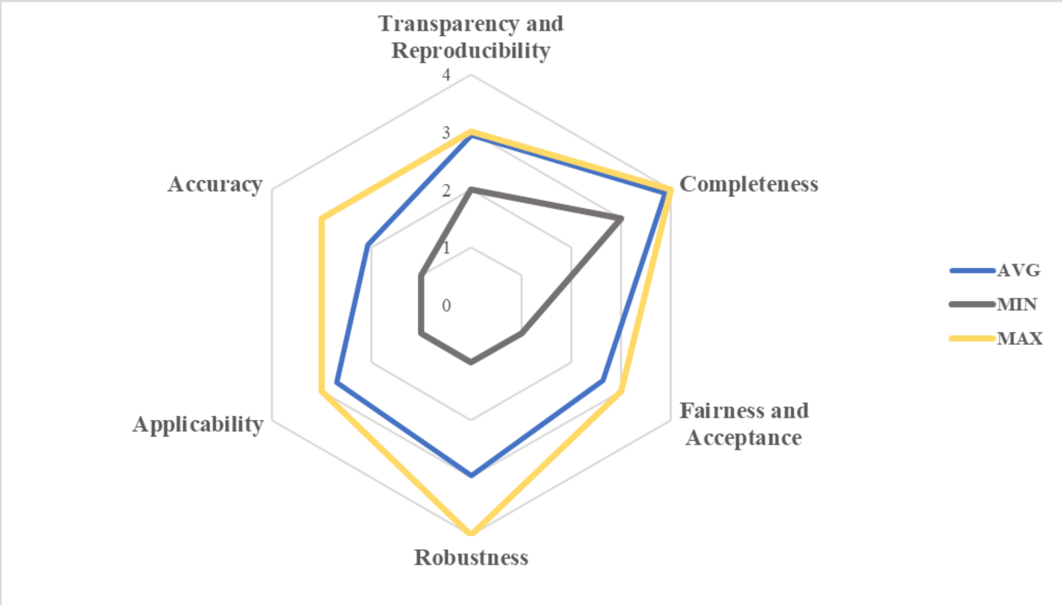


Figure 2: General criteria and Specific (Accuracy) criterion average scores assigned to the different papers evaluated

For the estimation of accuracy, there was a mean value of 2.14 for the eleven different identified methods. The highest value of accuracy was scored for direct measurement of enteric emissions using a respiratory chamber, although the reproducibility of the value obtained is strongly influenced by the environmental and experimental conditions (Hammond et al., 2016).

Out of the nine papers dealing with cattle, there were four focused on beef cattle, whilst three others involved dairy cattle or/and other species, and two involved beef cattle connected with wider aspects of LCA and involving other livestock species. Surveys considered California, Canadian, Northern Italy, Norway, and UK beef production systems, the case study considered Paraguayan beef production system, and methodological papers considered UK beef production systems and a comparison among different scenarios of Australian and Canadian beef farms. We identified ten methods used for beef sector enteric emission estimation, with 50% of them represented by the IPCC 2006 Tier 2 approach followed by the 30% of adapted IPCC Tier 2 approach and specific equations based on dietary composition of the dry matter intake (Ellis et al., 2007; Moares et al., 2014). The mean value of accuracy of the methods used was 2.12.

Small ruminants were included in four LCA papers, alone (three papers), and in combination with other species (one paper). Two papers were surveys, one showed a comparison between different raising systems, and one was a case study. Four were the methods utilised to estimate the enteric emissions: IPCC Tier 2 was utilised in one paper while in two papers an adapted IPCC Tier 2 method was used, and IPCC Tier 1 was used once, and the mean value of accuracy was 1.67.

Pigs were involved in two methodological papers and enteric emissions were evaluated according to Tier 1, while poultry was not considered in the selected papers.

DESCRIPTION OF KEY IDENTIFIED METHODOLOGIES

Within this section a brief description of each identified LCA methodology is presented. In this research, the different methods are discussed following a Tier approach as proposed by the IPCC. The FAO LEAP framework proposes three tiers to distinguish the level of complexity: Management (simple empirical) based models (Tier 1); Basic process or complex empirical models (Tier 2); Complex simulation models and direct measurement (Tier 3)(FAO, 2020). Direct observations generally belong to Tier 3 methods, while simple emission factors specific to large geographical areas belong to Tier 1, as discussed in previous research (Goglio et al., 2015).

Management (simple empirical) based models (Tier 1)

IPCC Tier I method (2006): this approach follows a simplified method that relies on default emissions factors, multiplied for the average annual number of animals in each herd category. The objective of the method is to estimate the enteric emissions of a herd or herd category at local or country level. Consequently, the main constraint of this method is the correct estimation of the number of animals raised on the study area or country, based on data available at different scale (local, regional, or national level) and, when not available, through the Food and Agriculture Organisation database. An advanced IPCC Tier 1 method is the IPCC **Tier 1a.** It is applicable when low and high productivity systems coexist in the same country. In this case, different emissions factors are considered both for high and low yielding herds (Dong et al., 2006).

IPCC Tier 1 method (2019): this newer approach follows updated guidelines and emission factors defined for a range of livestock species and management systems. For the latter, specific emission factors for each geographical region and livestock system have been developed (Gavrilova et al., 2019).

Basic process or complex empirical models (Tier 2)

IPCC Tier 2 method (2006): this method is based on a more complex approach that takes in consideration detailed data on gross energy intake and methane conversion factors for different livestock categories (Dong et al., 2006). IPCC Tier 2 method should be used if enteric fermentation is considered a key source for each livestock category and represents a large proportion of the total emissions. The key information which the method needs are: *i)* the number of animals in each considered category, *ii)* the daily gross energy intake, *iii)* the methane conversion factor expressed as the percentage of daily gross energy intake (Ym) lost as methane. The IPCC Tier 2 methodology (2006) method does not consider the following aspects to estimate the methane conversion factor: *i)* effects of digestibility, *ii)* diet dry matter intake or gross energy intake as it relates to live body weight, *iii)* diet chemical composition, *iv)* particle passage and digestion kinetics, *v)* plant microbial defensive compounds, *vi*) and variation in the microbial populations within the digestive tract (Dong et al., 2006).

IPCC Tier 2 method (2019): the updated IPCC Tier 2 method follows a more complex approach which requires detailed animal-specific data on gross energy intake and methane conversion factors (Ym) for specific livestock groups. The methodology provides emission factors for livestock categories taking account of the management systems and diet type. For other livestock categories updated factors have been also provided (Gavrilova et al., 2019).

Complex simulation models and direct measurements (Tier 3)

These types of models aim at considering in more detail the physiological digestive process and the interaction between animal physiological/productive level and enteric emission to provide the most accurate representation of the digestive process, based on data obtained by testing regression equations or direct measurements made using respiratory chambers. Whilst the most accurate, these types of models are also the most demanding for data variables, and usually require a thorough calibration to achieve satisfactory results (Rotz, 2018).

IPCC Tier 3 method (2006): Tier 3 methods (2019) are often used when livestock emissions are a considerable part of the total GHG emissions at country, local region or production systems level. This approach is required when it is necessary to go beyond the Tier 2 method by taking into account additional country or specific production system information. This method requires the development of complex models that consider the diet composition in detail, seasonal variation in animal population or feed availability or nutritional characteristics. Many of these estimated parameters are obtained by direct experimental evaluations (Dong et al., 2006; Gavrilova et al., 2019).

Yan method: this approach follows a dietary composition-based method based on the following key parameters: digestible energy intake, acid deterged fibre intake, silage dry matter intake, total dry matter intake and the feeding level above maintenance (Yan et al., 2000). A set of equations based on these parameters, have been set up for dairy and beef livestock systems (Yan et al., 2000).

Ellis method: a dietary composition method based on a set of equations considering several drivers in methane emissions for dairy and beef systems. The drivers considered include dry matter intake, neutrally deterged fibre intake (NDF), acid detergent fibre intake (ADF), forage proportion, and lignin (Ellis et al., 2007).

Moraes method: this method is based on a set of equations which relates the dietary composition to the methane emissions in dairy and beef livestock systems. Key parameters considered in the equations are energy intake, dietary fibre and lipid proportions, animal body weight and milk fat proportion, which were identified as key explanatory variables for predicting emissions (Moraes et al., 2014).

Niu method: this approach is based on an inter-continental database for dairy cow enteric emissions, and prediction models. The authors of this method concluded that information regarding dry matter intake was essential, with other factors such as neutral detergent fibre (NDF) improving the predictive power (Niu et al., 2018).

Belanche methods: within this method, enteric methane prediction models for sheep production were developed within an inter-continental database. Dry matter intake was found to be the most relevant variable, whilst factors such as age improved the prediction accuracy. The universal equations were found to have greater accuracy than the existing IPCC (2019) equations.

Estimation methods based on direct measurements: the direct measurement of enteric emissions using respiratory chambers improves strongly the accuracy of results provided, but on the other side, it does not guarantee a good reproducibility of the results obtained because of the variation of environmental conditions and animal management (Grandl et al., 2016). Other methods of direct measurement may also be used, such as greenfeed, SF6, Sniffer, laser gun or hoods or masks for example, but these are generally seen as less robust, e.g. for establishing IPCC emission factors, due to the specific circumstances that they are used in, which may not result in values which are applicable elsewhere.

Discussion

IDENTIFIED KEY METHODOLOGICAL ISSUES.

The Tier methodology used by the IPCC is widely adopted in both in dairy and beef cattle enteric emissions evaluation, being found within 50% of the papers evaluated. The level of accuracy achieved by the Tier methods is growing according to the complexity of the different parameters considered ranging between the value 1 for the IPCC Tier 1 method to the value 2.5 for improved Tier 2 (Alvarez-Hess et al., 2019; Salvador et al., 2017) and IPCC Tier 3 (Boxmeer et al., 2021; Klootwijk et al., 2016; Mostert et al., 2019) methods which require more detailed information about diet composition and category of raised animals.

The IPCC Tier 2 method assumes that on average 6.5%±1% and 3%±1% of Gross Energy Intake (GEI) is converted to CH_4 in dairy and beef cattle fed forage-based diets and in feedlot beef cattle diets with more than 90% of Dry Matter Intake (DMI) concentrates, respectively. The specific Ym values are then further detailed depending on the type of production system. Concerning high forage beef cattle diets, IPCC Tier 2 shows good statistical performance in terms of model efficiency despite its simplicity as reported by Escobar-Bahamondes et al., (2016). Nevertheless, the same authors identified evidence that the results achieved show higher random error when compared with more complex equations which consider diet composition, reflecting the Ym utilized (6.5%), as consequence of greater evidence from diets rich in forage compared to those with greater concentrate feed use. The same authors highlighted that the CH_4 GEI-based conversion factor likely overestimates the emissions in cows fed a diet with a medium proportion of forage. In the same paper, the results obtained by Escobar-Bahamondes et al., (2016) were consistent with Ricci et al., (2013) who reported that the IPCC (2006) default equations overestimate CH_4 by 26%, when compared with equations developed for cattle using a diets database (Dong et al., 2006).

In a more recent paper, Benaouda et al., (2019) showed the best enteric fermentation estimates are obtained when DMI, GEI or the feeding level (DMI/BW) are used as predictor variables. Nevertheless, at higher DMI, the best emissions predictions were associated with nonlinear equations as proposed by Mills et al., (2003) and Ramin and Huhtanen, (2013). The curvilinear effect is due to the higher rumen outflow, because of the use of a large proportion of starchy concentrates, which determines a shift in fermentation pattern from acetogenic to propionic at high DMIs. Concerning beef cattle, Benaouda et al., (2019) also stressed the need to develop new models.

Enteric emission evaluations obtained for small ruminants are based on IPCC methods, which are the most used, and papers devoted to the evaluation of methods are few and identify a relatively high correlation with empirical data using e.g. Root Mean Squared Prediction Error (RMSPE), (Dong et al., 2006; Gavrilova et al., 2019). The value of RMSPE was apparent when the IPCC 2006 method was used to evaluate sheep CH_4; where RMSPE values ranged from 23.1% (Patra et al., (2016); n=98) to 30% (Benaouda et al., (2019); n=111) when a meta-analysis was performed. The equation proposed by Patra et al., (2016), which utilised Digestible Energy Intake (DEI, MJ/d) as the predictor factor, showed a higher correlation between observed and predicted values. Belanche et al., (2023) found that increasing model complexity added accuracy in sheep equations. Dry matter intake was the primary predictor, with other factors including age, digestibility or proportion propionate increasing accuracy. Use of more livestock specific variables improved accuracy beyond that of the IPCC 2019 equations.

For pigs, the IPCC Tier 1 approach is still considered as the main method used to evaluate their enteric emissions (Dong et al., 2006; Gavrilova et al., 2019), although only one paper considered this species.

RESEARCH NEED, FUTURE STUDIES

In this study, it was observed that the Tier methodologies proposed by the IPCC can be considered satisfactory in that they are relatively simple and limit the heterogeneity of the results, in comparison with more complex models, as previously reported for soil C and LCA of cropping systems (FAO, 2018; Goglio et al., 2015). However, there is a tendency for Tier 1 approaches to overestimate emissions due to the use of factors that are too unrepresentative of all practices (mainly for not accounting for the nutritional quality of the diet), as was reported for soil N_2O emissions and for LCAs of agricultural systems (Aguilera et al., 2014; Nemecek et al., 2014; Rochette et al., 2018). Therefore, if such simplified approaches can be preferred for the reasons cited, the associated emission factors should be either more specific or derived from broader individual data, encompassing more representative sampling. Based on the results of this review, conversion factor adjustments would be expected to perform better in broader situations regarding feed diets. While certain IPCC Tier 2 parameters have been refined from the 2006 IPCC guidelines (Dong et al., 2006), including the methane conversion rate (Ym) for cattle (dairy and non-dairy) and buffalo, which now vary based on animal diet and level of productivity, a constant effort must be made to refine this conversion factor. For cattle, the refinement now describes four methane conversion factors (Ym) depending on productivity level and both feed quality digestibility (DE) and Neutral Detergent Fiber (NDF, DMI) ranging from 5.7 to 6.5 for dairy cattle and from 3 to 6.3 for non-dairy cattle (Gavrilova et al., 2019). The main concern is that different models produce different methane conversion rates thus both an identification of causes of variation of emissions and an additional guidance on model application are needed. For non-cattle species, the IPCC guidelines remain simple, and equations such as those within Belanche e al., (2023) would be preferable.

LCA METHODOLOGICAL GUIDELINES/RECOMMENDATIONS

From the analysis of the current LCA methods, some general recommendations can be made regarding the suitability and application of methods when undertaking a livestock system LCA.
For the estimation of enteric fermentation emissions for the purposes of an LCA, direct observations with specific devices or measurement within a metabolic chamber are preferable. However, when these facilities are unavailable, it is recommended to apply the IPCC 2019 Tier 2 methodology for its wide applicability (Gavrilova et al., 2019). Other equations can be applied that may be more specific to the feeding situation, e.g. based on Niu et al., (2018) for dairy cattle, Van Lingen et al. (2019) for beef cattle or for sheep (e.g. Belanche et al., (2023), however, when non-IPCC methods are used, then limitations should be highlighted and discussed, as suggested by the LCA ISO standards (ISO, 2020a, 2020b).

Conclusion

Following the analysis of the available literature, a series of recommendations were proposed. Where possible, and with data availability, more complex methods should be adopted for greater accuracy. More complex emission factor equations have been conceived for enteric fermentation, whilst at the other end of

the complexity spectrum, the IPCC Tier 1 methodology has been employed in most of the assessments analysed here. Independently of the method adopted, limitations should be discussed.
Future developments of the LCA methodology are necessary to improve LCAs of livestock systems. For enteric fermentation emissions, new inter-continental databases are providing improved accuracy. However, further research in developing a basic process model which results as a compromise between applicability and accuracy is desirable, and emission factors should better reflect herd characteristics and livestock management depending on the LCA objectives. This LCA method development must be synchronous with improvements in observation methods and the assessment of different crop-livestock management systems.

References

Aguilera, E., Guzmán, G., Alonso, A., 2014. Greenhouse gas emissions from conventional and organic cropping systems in Spain. I. Herbaceous crops. Agron. Sustain. Dev. 35, 713–724 doi: 10.1007/s13593-014-0267-9.

Alvarez-Hess, P.S., Little, S.M., Moate, P.J., Jacobs, J.L., Beauchemin, K.A., Eckard, R.J., 2019. A partial life cycle assessment of the greenhouse gas mitigation potential of feeding 3-nitrooxypropanol and nitrate to cattle. Agricultural Systems 169, 14–23 doi: 10.1016/j.agsy.2018.11.008.

Belanche, A., Hristov, A.N., van Lingen, H.J., Denman, S.E., Kebreab, E., Schwarm, A., Kreuzer, M., Niu, M., Eugène, M., Niderkorn, V., Martin, C., Archimède, H., McGee, M., Reynolds, C.K., Crompton, L.A., Bayat, A.R., Yu, Z., Bannink, A., Dijkstra, J., Chaves, A.V., Clark, H., Muetzel, S., Lind, V., Moorby, J.M., Rooke, J.A., Aubry, A., Antezana, W., Wang, M., Hegarty, R., Hutton Oddy, V., Hill, J., Vercoe, P.E., Savian, J.V., Abdalla, A.L., Soltan, Y.A., Gomes Monteiro, A.L., Ku-Vera, J.C., Jaurena, G., Gómez-Bravo, C.A., Mayorga, O.L., Congio, G.F.S., Yáñez-Ruiz, D.R., 2023. Prediction of enteric methane emissions by sheep using an intercontinental database. J. Clean. Prod. 384, 135523 doi: 10.1016/j.jclepro.2022.135523.

Benaouda, M., Martin, C., Li, X., Kebreab, E., Hristov, A.N., Yu, Z., Yáñez-Ruiz, D.R., Reynolds, C.K., Crompton, L.A., Dijkstra, J., Bannink, A., Schwarm, A., Kreuzer, M., McGee, M., Lund, P., Hellwing, A.L.F., Weisbjerg, M.R., Moate, P.J., Bayat, A.R., Shingfield, K.J., Peiren, N., Eugène, M., 2019. Evaluation of the performance of existing mathematical models predicting enteric methane emissions from ruminants: Animal categories and dietary mitigation strategies. Animal Feed Science and Technology 255, 114207 doi: https://doi.org/10.1016/j.anifeedsci.2019.114207.

Boxmeer, E. van, Modernel, P., Viets, T., 2021. Environmental and economic performance of Dutch dairy farms on peat soil. Agricultural Systems 193, 103243 doi: https://doi.org/10.1016/j.agsy.2021.103243.

Brady, N., Weil, R., 2002.The Nature and Properties of Soils, 13th ed. Prentice Hall, Upper Saddle River, New Jersey, USA.

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. Anim. 7, 212–219 doi: 10.1017/S1751731113000785.

Don, A., Seidel, F., Leifeld, J., Kätterer, T., Martin, M., Pellerin, S., Emde, D., Seitz, D., Chenu, C., 2024. Carbon sequestration in soils and climate change mitigation—Definitions and pitfalls. Global Change Biology 30, e16983 doi: 10.1111/gcb.16983.

Dong, H., Mangino, J., McAllister, T.A., Hatfield, J.L., Johnson, D.E., Lassey, K.R., Lima, M.A., Romanovskaya, A., Bartram, D., Gibb, D., Martin, J.H., Jr, 2006.Chapter 10: Emissions from Livestock and Manure Management, IPCC guidelines for national inventories. Intergovernmental Panel for Climate Change (IPCC), Geneva.

Ellis, J.L., Kebreab, E., Odongo, N.E., McBride, B.W., Okine, E.K., France, J., 2007. Prediction of Methane Production from Dairy and Beef Cattle. Journal of Dairy Science 90, 3456–3466 doi: 10.3168/jds.2006-675.

Escobar-Bahamondes, P., Oba, M., Beauchemin, K., 2016. Universally applicable methane prediction equations for beef cattle fed high- or low-forage diets. Can. J. Anim. Sci. CJAS-2016-0042 doi: 10.1139/CJAS-2016-0042.

FAO, 2016a.Environmental Performance of Pig Supply Chains: Guidelines for assessment (Livestock Environmental 251 Assessment and Performance Partnership). Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016b.Environmental performance of animal feeds supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016c.Greenhouse gas emissions and fossil energy use from small ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016d.Greenhouse gas emissions and fossil energy use from poultry supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2016e.Environmental performance of large ruminant supply chains: Guidelines for assessment. Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2017.Livestock solutions for climate change. Food and Agriculture Organisation of the United Nations, Rome, Italy.

FAO, 2018.Measuring and modelling soil carbon stocks and stock changes in livestock production systems – Guidelines for assessment (Draft for public review). Livestock Environmental Assessment and Performance (LEAP) Partnership. Food and Agriculture Organization of the United Nations, Rome.

FAO, 2020.Livestock Environmental Assessment and Performance (LEAP) Partnership | Food and Agriculture Organization of the United Nations. http://www.fao.org/partnerships/leap/en/ (accessed 11 may 2020).
Gavrilova, O., Leip, A., Dong, H., MacDonald, J., Gomez Bravo, C., Amon, B., Barahona Rosales, R., Del Prado, A., De Lima, M., Oyhantçabal, W., Van Der Weerden, T., Widiawati, Y., Bannink, A., Beauchemin, K., Clark, H., Leytem, A., Kebreadìb, E., Ngwabie, N., Imede Opio, C., Vanderzaag, A., Vellinga, T., 2019.2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Chapter 10: Emission from livestock and manure management. IPCC, Interngovernmental Panel for Climate Change, Geneva.
Godfray, H.C.J., Aveyard, P., Garnett, T., Hall, J.W., Key, T.J., Lorimer, J., Pierrehumbert, R.T., Scarborough, P., Springmann, M., Jebb, S.A., 2018. Meat consumption, health, and the environment. Sci. 361, eaam5324 doi: 10.1126/science.aam5324.
Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015. Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J. Clean. Prod. 104, 23–39 doi: 10.1016/j.jclepro.2015.05.040.
Goglio, P., Knudsen Trydeman, M., 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. Letters 4, 100035 doi: 10.1016/j.clpl.2023.100035.
Google, 2024.Google Forms – create and analyse surveys, for free. https://www.google.com/intl/en-GB/forms/about/ (accessed 13 january 2022).
Grandl, F., Amelchanka, S.L., Furger, M., Clauss, M., Zeitz, J.O., Kreuzer, M., Schwarm, A., 2016. Biological implications of longevity in dairy cows: 2. Changes in methane emissions and efficiency with age. Journal of Dairy Science 99, 3472–3485 doi: 10.3168/jds.2015-10262.
Grandl, F., Furger, M., Kreuzer, M., Zehetmeier, M., 2019. Impact of longevity on greenhouse gas emissions and profitability of individual dairy cows analysed with different system boundaries. animal 13, 198–208 doi: 10.1017/S175173111800112X.
Grossi, G., Goglio, P., Vitali, A., Williams, A.G., 2019. Livestock and climate change: impact of livestock on climate and mitigation strategies. Anim. Frontiers 9, 69–76 doi: 10.1093/af/vfy034.
IPCC, 2022. Climate change 2022: Mitigation of climate change. WGIII Mitigation of Climate Change Climate Change 2022 Working Group III contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, Interngovernmental panel for climate change, Geneva, Switzerland.
ISO, 2020a. EN ISO 14044:2006+A2:2020. Environmental Management – Life Cycle Assessment – Requirements and Guidelines. International Organization for Standardization, Geneva.

ISO, 2020b. EN ISO 14040:2006+A1 Environmental Management- Life Cycle Assessment, Principles and Framework. International Organization for Standardization, Geneva.

Jourdaine, M., Loubet, P., Trebucq, S., Sonnemann, G., 2020. A detailed quantitative comparison of the life cycle assessment of bottled wines using an original harmonization procedure. J. Clean. Prod. 250, 119472 doi: 10.1016/j.jclepro.2019.119472.

López-Andrés, J.J., Aguilar-Lasserre, A.A., Morales-Mendoza, L.F., Azzaro-Pantel, C., Pérez-Gallardo, J.R., Rico-Contreras, J.O., 2018. Environmental impact assessment of chicken meat production via an integrated methodology based on LCA, simulation and genetic algorithms. J. Clean. Prod. 174, 477–491 doi: 10.1016/j.jclepro.2017.10.307.

Macombe, C., Loeillet, D., Gillet, C., 2018. Extended community of peers and robustness of social LCA. Int. J. Life Cycle Assess. 23, 492–506 doi: 10.1007/s11367-016-1226-2.

Mills, J.A.N., Kebreab, E., Yates, C.M., Crompton, L.A., Cammell, S.B., Dhanoa, M.S., Agnew, R.E., France, J., 2003. Alternative approaches to predicting methane emissions from dairy cows1. Journal of Animal Science 81, 3141–3150 doi: 10.2527/2003.81123141x.

Moraes, L.E., Strathe, A.B., Fadel, J.G., Casper, D.P., Kebreab, E., 2014. Prediction of enteric methane emissions from cattle. Global Change Biology 20, 2140–2148 doi: 10.1111/gcb.12471.

Morris, J., Brown, S., Cotton, M., Matthews, H.S., 2017. Life-Cycle Assessment Harmonization and Soil Science Ranking Results on Food-Waste Management Methods. Environ. Sci. Technol. 51, 5360–5367 doi: 10.1021/acs.est.6b06115.

Mostert, P.F., Bokkers, E.A.M., Boer, I.J.M. de, Middelaar, C.E. van, 2019. Estimating the impact of clinical mastitis in dairy cows on greenhouse gas emissions using a dynamic stochastic simulation model: a case study. Animal 13, 2913–2921 doi: https://doi.org/10.1017/S1751731119001393.

Nemecek, T., Schnetzer, J., Reinhard, J., 2014. Updated and harmonised greenhouse gas emissions for crop inventories. Int. J. Life Cycle Assess. 1–18 doi: 10.1007/s11367-014-0712-7.

Patra, A.K., Lalhriatpuii, M., Debnath, B.C., 2016. Predicting enteric methane emission in sheep using linear and non-linear statistical models from dietary variables. Anim. Prod. Sci. 56, 574–584.

Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Sci. 360, 987–992 doi: 10.1126/science.aaq0216.

Ramin, M., Huhtanen, P., 2013. Development of equations for predicting methane emissions from ruminants. Journal of Dairy Science 96, 2476–2493 doi: https://doi.org/10.3168/jds.2012-6095.

Rotz, C.A., 2018. Modeling greenhouse gas emissions from dairy farms. J. Dairy Sci. 101, 6675–6690 doi: 10.3168/jds.2017-13272.

Rotz, C.A., Corson, M.S., Chianese, D.S., Montes, F., Hafner, S.D., Bonifacio, H.F., Coiner, C.U., 2018.Integrated Farm System Model: Reference Manual. USDA Agricultural Research Serv., University Park, PA, US.

Salvador, S., Corazzin, M., Romanzin, A., Bovolenta, S., 2017. Greenhouse gas balance of mountain dairy farms as affected by grassland carbon sequestration. Journal of Environmental Management 196, 644–650 doi: 10.1016/j.jenvman.2017.03.052.

Segura-Salazar, J., Lima, F.M., Tavares, L.M., 2019. Life Cycle Assessment in the minerals industry: Current practice, harmonization efforts, and potential improvement through the integration with process simulation. J. Clean. Prod. 232, 174–192 doi: 10.1016/j.jclepro.2019.05.318.

Siegert, M.-W., Lehmann, A., Emara, Y., Finkbeiner, M., 2019. Harmonized rules for future LCAs on pharmaceutical products and processes. Int. J. Life Cycle Assess. 24, 1040–1057 doi: 10.1007/s11367-018-1549-2.

Taki, R., Wagner-Riddle, C., Parkin, G., Gordon, R., VanderZaag, A., 2019. Comparison of two gap-filling techniques for nitrous oxide fluxes from agricultural soil. Can. J. Soil. Sci. 99, 12–24 doi: 10.1139/cjss-2018-0041.

Van Lingen, H.J., Niu, M., Kebreab, E., Valadares Filho, S.C., Rooke, J.A., Duthie, C.-A., Schwarm, A., Kreuzer, M., Hynd, P.I., Caetano, M., Eugène, M., Martin, C., McGee, M., O'Kiely, P., Hünerberg, M., McAllister, T.A., Berchielli, T.T., Messana, J.D., Peiren, N., Chaves, A.V., Charmley, E., Cole, N.A., Hales, K.E., Lee, S.-S., Berndt, A., Reynolds, C.K., Crompton, L.A., Bayat, A.-R., Yáñez-Ruiz, D.R., Yu, Z., Bannink, A., Dijkstra, J., Casper, D.P., Hristov, A.N., 2019. Prediction of enteric methane production, yield and intensity of beef cattle using an intercontinental database. Agr. Ecosyst. Environ. 283, 106575. doi: 10.1016/j.agee.2019.106575

Yan, T., Agnew, R.E., Gordon, F.J., Porter, M.G., 2000. Prediction of methane energy output in dairy and beef cattle offered grass silage-based diets. Livestock Production Science 64, 253–263 doi: 10.1016/S0301-6226(99)00145-1.

Yan, T., Frost, J.P., Agnew, R.E., Binnie, R.C., Mayne, C.S., 2006. Relationships Among Manure Nitrogen Output and Dietary and Animal Factors in Lactating Dairy Cows. Journal of Dairy Science 89, 3981–3991 doi: 10.3168/jds.S0022-0302(06)72441-9.

Zampori, L., Pant, R., 2019.Suggestions for updating the Product Environmental Footprint (PEF) method. EUR 29682 EN, Publications Office of the European Union, Luxembourg.

SG6: Methodology of Social Life Cycle Assessment (S-LCA) of Livestock Value Chains in Europe

Annabel Oosterwijk[a] , Seval Cicek[a], Pietro Goglio[b], Coen van Wagenberg[a]

[a]Wageningen Social and Economic Research, Droevendaalsesteeg 4, 6708 PB, Wageningen, The Netherlands

[b]Department of Agricultural, Food and Environmental Sciences, Borgo XX Giugno 74, 06124 Perugia (PG), Italy

Introduction

The transition towards a more sustainable food systems is one of the great challenges at global level. While environmental considerations have been widely explored, the social sustainability of agri-food systems has been scarcely addressed in literature (Mancini et al., 2022).

Increasingly more focus is given to social impact assessment of both agrifood and livestock system in the corporate reporting standards directives both at European and global level (UN Sustainable Development Goals, CSRD, etc.). In addition, a growing number of people assesses product quality not just by intrinsic attributes but also by extrinsic attributes connected with sustainability (Zira, 2020).

Livestock systems are crucial in societies worldwide, serving various roles like providing food, income, and cultural significance. They also offer ecosystem benefits and fulfil owners' emotional needs. Besides a positive impact, animal farming can also have a negative impact on the sustainability of food production and society (Busch, 2023). Across the EU, the livestock sector plays a significant economic and social role. For instance, in 2017, the value of livestock production and livestock products in the EU-28 was equal to € 170 billion. Furthermore, European livestock farms employ around 4 million people, with on average, 1 to 2 workers per livestock farm. In terms of consumption, protein of animal origin covers over 50% of the total protein content of European diets (Directorate - General for Agriculture and Rural Development, 2020). Agriculture (including livestock) and forestry rank among Europe's riskiest professions due to frequent accidents, jeopardizing sector sustainability. Over the past decade, an average of 500+ deaths annually and 150,000+ non-fatal accidents occurred. A 2012 EU survey found that agricultural workers rated their job's impact on their health higher than workers in any other industry (OSHA Europa, 2022). Yet, despite its size and impact on society, social life cycle assessment studies in agriculture and as well as in livestock production systems are very limited.

There is a growing need for actors to address not only environmental aspects but also social sustainability aspect of the European livestock system. The complexity of social issues and a lack of data makes this task challenging. Yet, there are methodologies developed in conducting social impact assessment that are increasingly being applied and improved.

A useful methodology for assessing social impacts from a product perspective is Social Life Cycle Assessment (S-LCA). The S-LCA has been standardized in the guidelines for Social Life Cycle Assessment of a Product (UNEP 2020). In the context of S-LCA, social impacts are consequences of positive or negative pressures on social areas of protection (i.e., well-being of stakeholders) while a social hotspot is defined as the location and/or activity in the life cycle where a social issue (negative impact or benefit) and/or social risk is likely to occur. S-LCA can be applied at micro- (product and/or company), meso- (economic sector or region), and macro- (country, state) levels (Mancini et al., 2022).

The S-LCA methodology has a lower level of methodological maturity and implementation compared to environmental LCA (E-LCA). While the UNEP Guidelines (2020) provide new guidance on the impact assessment phase in an S-LCA, normalization and weighting still lack a common reference, e.g. in terms of normalization factors and a conventional weighting scheme (Mancini et al., 2022).

In the PATHWAYS project, Social Life Cycle Assessment (S-LCA) is applied to address social sustainability considerations of European livestock production systems. The integrated assessment of the social dimension of sustainability from a life cycle perspective is relying on S-LCA. S-LCA aims at assessing the social impacts of products and services across their life cycle, from extraction of raw material to the end-of-life phase (UNEP, 2020).

This deliverable focuses on the S-LCA methodology and the way it can be operationalized to assess the social footprint of European livestock systems within the PATHWAYS project. Chapter 1 addresses the general S-LCA methodology and its various approaches. The methodology and approach which does align the best with the PATHWAYS project is further examined in chapter 2. Later in the project, we will apply this S-LCA methodology and approach to case studies and livestock systems selected in the PATHWAYS project.

S-LCA in general

In general, S-LCA consists of four phases: 1) setting the Goal and Scope (G&S) of the study, 2) collecting data (Life Cycle Inventory), 3) assessing the risks and potential impacts (Impact Assessment), and 4) interpreting results (Interpretation). Because different S-LCA methods have diverging purposes and applications, this chapter describes the basic principles of the four phases and the methods. The same principles and steps will be also applied for the application of S-LCA within the PATHWAYS project.

GOAL AND SCOPE OF THE STUDY

In this first phase of the S-LCA the purpose and the methodological framework of the study are determined.

GOAL DEFINITION

The goal of a S-LCA study specifies why the study is conducted. The goal can be, for example, to get more insights into the social impact of a product/service, to examine potential social improvement options along the life cycle, to identify hotspots of a product and/or organization, or to quantify the social performance of a product/service (UNEP, 2020).
In the goal definition, the target audience also needs to be defined. It needs to be determined whether the study is intended for internal or external use. Ideally, the goal of the study also specifies whether to align it with attributional or consequential thinking, which will impact other methodological choices (UNEP, 2020).

SCOPE DEFINITION

In the scope definition phase, the functional unit, reference flow, product system, system boundaries, activity variable, stakeholder categorization, impact (sub)categories and performance indicators are defined among others. All these items are in detail explained in UNEP guidelines for S-LCA (UNEP, 2020).
The functional unit defines quantitatively the object of a study, for example 1 kg of beef. However, it is not always possible to link all social indicators to the product in a quantitative way. The indicators can be divided in functional unit-related and non-functional unit-related indicators, as done by Chen (2017). The reference flow, product system and system boundaries are defined the same way as in E-LCA. An activity variable is used as a measure of process activity which can be related to process output (UNEP, 2020). They reflect the share of a given activity associated with each unit process.

Determination of stakeholder categories and impact (sub)categories

A stakeholder category is a group type which can be affected by the activities or organizations involved in the life cycle of a product, service or organization under consideration. The UNEP (2020) guidelines include the following stakeholder categories: workers, local communities, value chain actors, consumers, society and children. The selection of stakeholder categories also affects the choice of impact categories and subcategories at each step of the life cycle.
The main rule is that all relevant stakeholders and impact categories should be considered in a S-LCA study (UNEP, 2020). There is a wide variety of impact (sub)categories and performance indicators adopted for assessing social impacts for each stakeholder category. The selection of the impact (sub)categories depends

on the method, data availability and specific contexts. Table 1 presents the impact (sub)categories by stakeholder category which are included in the UNEP Guidelines (2020).

Table 1. List of impact (sub)categories by stakeholder category as presented in the UNEP Guidelines (2020)

Stakeholder category					
Worker	Local community	Value chain actors (excl. consumers)	Consumer	Society	Children
•Freedom of association and collective bargaining •Child labour Fair salary •Working hours •Forced labour •Equal opportunities / discrimination •Health and safety •Social benefits / social security •Employment relationship •Sexual harassment •Smallholders including farmers	•Access to material resources •Access to immaterial resources •Delocalization and migration •Cultural heritage •Safe and healthy living conditions •Respect of indigenous rights •Community engagement •Local employment •Secure living conditions	•Fair competition •Promoting social responsibility •Supplier relationships •Respect of intellectual property rights •Wealth distribution	•Health and safety •Feedback mechanism •Consumer privacy •Transparency •End-of-life responsibility•	•Public commitments to sustainability issues •Contribution to economic development •Prevention and mitigation of armed conflicts •Technology development •Corruption •Ethical treatment of animals •Poverty alleviation	•Education provided in the local communities •Health issues for children as consumer •Children concerns regarding marketing practices

The final set of impact (sub)categories can be selected by using a top-down and/or bottom-up approach, as advised by Kruse et al. (2009). The top-down approach identifies impact (sub)categories which focus on internationally recognized societal values (ILO, Human rights) whereas the bottom-up approach identifies impact (sub)categories on (but should not be limited to) industry or stakeholder interests and/or data availability (Kruse et al., 2009a). The top-down approach can be followed by a sectorial social risk analysis, which aims at completing the UNEP (2020) recommended list of impact (sub)categories through an extensive identification of social and socio-economic topics that are related to the studied sector. The bottom-up approach requires actors' consultation for the prioritization of relevant impact (sub)categories. Since socioeconomic impacts may vary between industries due to the nature of processes or products involved, this participatory approach is designed to gather information about the social significance of the list of impact (sub)categories from the top-down approach for directly affected and involved stakeholders, as defined by the S-LCA guidelines (UNEP, 2020).

The UNEP (2020) guidelines propose to use a participatory approach (i.e. an approach in which actors participate and contribute to the study or scientific process) to identify relevant stakeholder groups, impact (sub)categories and performance indicators. Applying participatory approaches in this selection considers the perspective and values of different stakeholders involved, increasing both the legitimacy and local relevance of the assessment (UNEP, 2020). Focus groups can be a type of group interview organized to acquire a portrait of a combined local perspective on a specific set of issues. Focus groups can also be used in impact assessment when defining the relative importance (weight) of each impact (sub)category.

Defining performance indicators

A performance indicator reflects the extent of the social impact and belongs to a certain impact (sub)category (UNEP, 2020). Impact (sub)categories are assessed with the use of performance indicators, of which inventory indicators link directly with the inventory of the product life cycle (UNEP, 2020). Several performance indicators may be used to assess each of the subcategories. These performance indicators may vary depending on the context of the study and the product analysed. An impact (sub)category may have various performance indicators, e.g., 'hours of missed education' and 'hours worked' are both performance indicators for the impact (sub)category 'child labour'. A participatory approach can be used in the selection of a final set of performance indicators. The status of impact or subcategories is assessed by collecting data on one or several performance indicators, selected to cover the most relevant aspects of the category. Table 2 gives an example of the linkage between stakeholders, impact (sub)category and performance indicator.

Table 2. Example of a linkage between stakeholders, subcategories, and performance indicators

Stakeholder	Impact (sub)category	Performance indicator
Worker	Occupational health and safety	Number/percentage of injuries or fatal accidents in the organization by job qualification inside the company
		The company or facility has conducted a health & safety assessment

LIFE CYCLE INVENTORY

In this second phase of S-LCA, all input and output flows are identified, as well as the social inventory performance indicators to be evaluated. The Social Life Cycle Inventory (S-LCI) is about collecting data for all unit processes within defined system boundaries. The S-LCI consists of the inventory of all flows of the studied system normalized per functional unit (e.g. 1 kg of meat) (UNEP, 2020). To obtain this inventory, we follow these steps (UNEP, 2020):

1.Divide the system under study into interconnected processes that provide products or services to one another, like fertilizer production and agricultural cultivation. This creates a flow chart, already part of the G&S.

2.Determine the flow amounts for each process, usually normalized to a process output. For example, it takes 5 kWh of electricity to produce 1 kg of fertilizer. Additionally, gather information about the system.

3.Quantify the total amounts of processes and their flows for the reference flow. This is typically done using a linear relationship. For instance, if 2 worker-hours are required for 1 kg of fertilizer, then 4 worker-hours are needed when 2 kg of fertilizer is required (activity variable).

4.Collect social inventory data for all processes and flows related to the main stakeholders (performance indicators) defined in G&S. In our example, this would include information such as the workers' salaries involved in producing 2 kg of fertilizer and 5 kWh of electricity.

DATA COLLECTION AND QUALITY

The data collection process in S-LCA can be time- and resource-consuming, particularly when gathering specific data from stakeholders for the impact (sub)categories included in the study (UNEP, 2020). Without prioritization, this would potentially require visiting a large number of sites. Hence, prioritization and estimating the relative importance of activities in a product system are crucial to guide data collection and allocate efforts effectively (UNEP, 2020).

There are several ways to prioritize data collection, such as literature review, data on activity variables and social hotspots. A first analysis can be conducted using a database and software to identify the social hotspots of the product system. Social hotspots are unit processes located in a region (e.g., a country) where a situation occurs which may be considered a problem, a risk, or an opportunity, in relation to a social issue that is threatening social well-being or that may contribute to its further development (UNEP, 2020). This social hotspot analysis can form the core of data collection prioritization in the S-LCA study and can be complemented with other data sources for some of the processes and made more specific over time in an iterative fashion (UNEP, 2020). The same principle is to be applied for the purpose of this study in the PATHWAYS project.

Activity variables

Worker-hours is the most used activity variable. It consists of the number of worker-hours necessary to complete a production activity/unit process. There are several approaches for collecting data on activity variables, such as site-specific data collection, the use of a S-LCA dedicated database (e.g., SHDB or PSILCA), and through input-output databases (e.g., GTAP) (UNEP, 2020). Many studies that include an activity variable use of S-LCA databases, which by default integrates the calculation of activity variable data (UNEP, 2020).

Data collection

Generally, data collection for impact assessment in S-LCA is comparable for the two types of impact assessment approaches (RS S-LCIA and IP S-LCIA) (UNEP, 2020), which are explained in more detail in Chapter " Social life cycle Impact assessment approaches" . Data are collected at the company and product level for the stakeholder groups and subcategories (RS S-LCIA) or impact categories (IP S-LCIA), as defined in the G&S of the study (UNEP, 2020). In all cases the collected data relate to the life cycle stages as defined in the product system. Site-specific and/or generic data as well as quantitative and/or qualitative data may be used depending on the requirements resulting from the definition of the G&S phase, see Figure 1.
For each of the impact (sub)categories selected and to be covered in a study in accordance with the G&S section, it is necessary to identify corresponding inventory performance indicators. These performance indicators should be compatible with the selected approach of impact assessment. Social inventory performance indicators (or social flows) are usually defined as simple variables (e.g., salary, number of accidents at workplace) providing the status of a certain topic/life cycle stage/process (Vanclay, 2002). They provide the most direct evidence of a social condition. The choice of social inventory performance indicators will determine the data which ought to be collected. In S-LCA, performance indicators can be of qualitative,

semi-quantitative, or quantitative nature. They can also be company specific, site-specific, generic, primary, or secondary (UNEP, 2020).

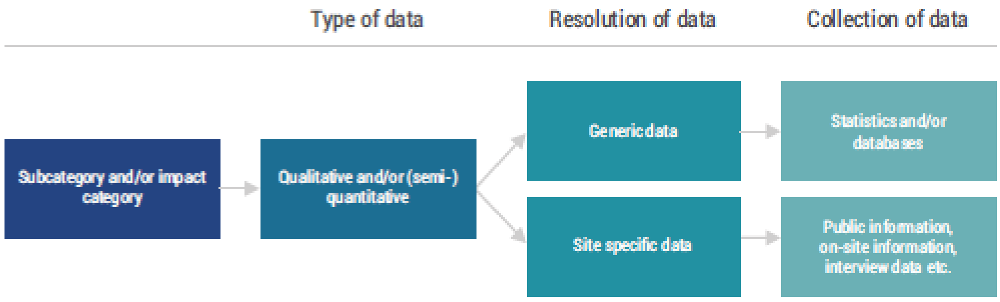


Figure 1. Data collection and interrelations in S-LCA (UNEP, 2020)

Typical sources of data for S-LCA comprise interviews, surveys, audit results, scientific and grey literature publications, generic databases, and others (UNEP, 2020). Each of these demands different levels of involvement in terms of methods and time from the practitioner. Therefore, depending on the goal of the study and the resources available, the strategy for data collection should be defined respectively.

Secondary data collection

Secondary data can be collected through a literature review, web search or through existing databases. This will depend on the data needs and level of detail required. Since S-LCA is an iterative procedure, a first analysis can be conducted using a database and software (e.g. PSILCA/SHDB database) to identify the social hotspots of the product system (UNEP, 2020). This generic analysis can form the core of the S-LCA and be complemented with other data sources for some of the processes (foreground or background) and made more specific over time in an iterative fashion (UNEP, 2020).

Primary data collection

The collection of primary data is carried out by visiting specific or relevant production sites or by working together with respective organizations. Thus, primary data can be gathered through direct contact with organizations and companies (e.g. by means of management systems), through NGOs or comparable organizations (e.g. by means of auditing processes), through observation of business/production processes on-site, or through interviews or surveys with affected stakeholders (e.g. workers or local inhabitants) (UNEP, 2020).

The need for primary data can be determined by starting with a first hotspot assessment using generic data and by identifying data gaps. Primary data are especially relevant for prioritized (foreground) processes and products. It is also relevant if the specific process or product performs better or worse than the defined

average based on the hotspot assessment. Furthermore, they are very relevant for measuring positive impacts, to determine their contribution to the specific product, plant, or company compared to the local condition. It is also necessary to collect primary data to verify the risk and to be able to analyse impacts. It is possible that some of the hotspots identified in the generic analysis end up not representing any problem in the production chain. On the other hand, problems can still appear where generic analysis did not suspect them (UNEP, 2020). Site-specific data are being collected through a range of methods, for example document auditing, interviews, questionnaires, participatory evaluation, etc.
The data collection strategy can be refined due to new knowledge, such as processes that are important and significant, significant topics and processes based on the social hotspots, unavailability of data and subsequent sensitivity analysis.

Data quality

Data quality needs to be addressed as it is fundamental to ensure the reliability and validity of the findings and to reach useful conclusions (UNEP, 2020). Currently, there is still no comprehensive guidance document addressing general data quality requirements and management for social and socio-economic data in S-LCA. Due to this, some general considerations and possible data quality management options are presented in the UNEP Guidelines 2020 where they can be taken as reference (UNEP, 2020. p 75-78).

Life Cycle Impact Assessment

In this third phase of a S-LCA, the Social Life Cycle Impact Assessment (S-LCIA), the aim is to calculate, understand and evaluate the magnitude and significance of the potential social impacts of a product system throughout the life cycle of the product.

It is important to note that S-LCIA mainly focuses on evaluating potential social impacts – not social impacts per se. As a reminder, potential social impact is understood as the likely presence of a social impact, resulting from the activities/behaviours of organizations linked to the life cycle of the product or service and for the use of the product itself (UNEP, 2020). The term “potential” is important as it conveys relativism. The assessment of potential impacts is supported by a range of hypotheses that, while being rigorous, have their own limitations.

Social life cycle Impact assessment approaches

S-LCIA approaches are classified into two main approaches: Reference Scale approach (RS S-LCIA/Type 1) and Impact Pathway approach (IP S-LCIA/Type 2), also depicted in Figure 2.

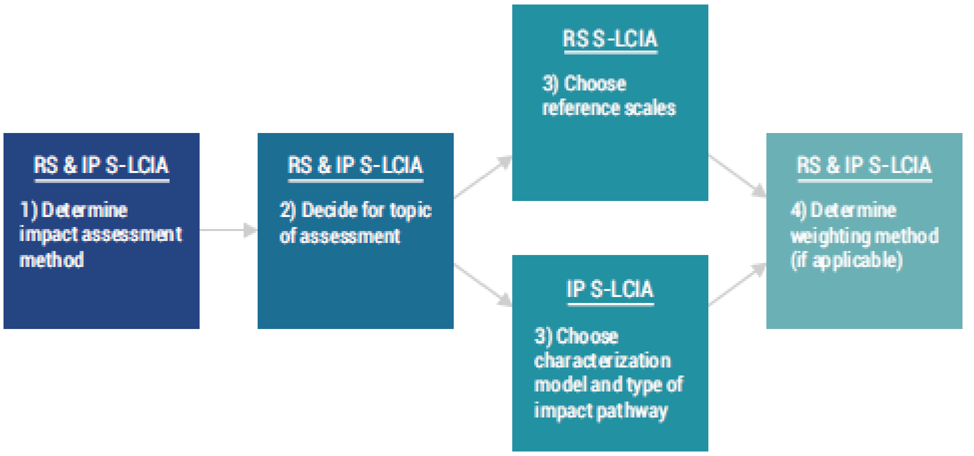


Figure 2. Two main approaches in S-LCA (UNEP, 2020)

The RS S-LCIA approach can be used in case the aim is to describe a product system with a focus on its social performance or social risk (UNEP, 2020). This social performance is based on specific reference points. The RS S-LCIA approach does estimate the magnitude and significance of potential social impacts (UNEP, 2020). In a RS S-LCIA data must be collected for creating the reference scales for the different stakeholder groups and the different subcategories identified as relevant for the study and (optional) for applying the activity variable or a weighting step.

The IP S-LCIA approach helps to predict the consequences of the product system, with an emphasis on characterizing potential social impacts (UNEP, 2020). This is done by assessing the potential or actual social impacts by using causal or correlation/regression-based directional relationships between the product system and the resulting potential social impacts (characterization) (UNEP, 2020). In an IP S-LCIA, data must be collected for all inventory indicators relevant to express the impact (sub)categories identified, for the characterization factors, and (optional) for applying the activity variable or a weighting step (UNEP, 2020).

The two approaches are distinct and did not experience the same history and are currently not at the same level of development and implementation (UNEP, 2020). While relatively young, RS S-LCIA approaches are operational in practice at present and numerous practical case studies exist. Meanwhile, studies applying IP S-LCIA approaches chiefly pertain to the field of research, but several documented pathways are available and readily applicable. The choice of which S-LCIA approach will be used influences the other methodological choices in the S-LCA. Table 3 provides some advantages and disadvantages of both approaches.

Table 3. Advantages and disadvantages of the Reference Scale (RS) social LCA and the Impact Pathways (IP) social LCA

Type of methodological S-LCA Approach	Advantages	Disadvantages
RS-S-LCA	• More developed than the IP-S-LCa • Larger data availability for most economic sectors • More reference data for comparison • It can be used for feasibility assessments	• Data mostly at national level • Not specific to process activities • Scale based approach
IP-S-LCA	• helps to predict the consequences of the product systems • better characterizing potential social impacts	• limited development of the methods • limited data availability

INTERPRETATION

The interpretation phase is built upon the requirements of ISO 14044 and it consists of the following steps (UNEP, 2020), also depicted in Figure 3.

Completeness check

Data completeness refers to an indication of whether all the data necessary to conduct the assessment is available.

Sensitivity and data quality check

Sensitivity analysis is a technique to assess whether a change (e.g., the inclusion or exclusion of a unit process) to the system would change the result above a certain threshold (in quantitative sensitivity analysis

a 1% or 5% change is often regarded as a significant change). Sensitivity analysis may also be performed on qualitative data, essentially estimating if the inclusion of a process would affect the overall result.
Also, a check on data validity must be conducted during the process of data collection to confirm and provide evidence that the data quality requirements have been fulfilled for the intended application.

Consistency check

The consistency check aims to verify the appropriateness of modelling, collected data and of the methodological choices made during each life cycle stage according to the defined G&S.

Materiality assessment

The materiality assessment aims to identify significant social performances or impacts, risks, stakeholder categories, life cycle phases of processes, in accordance with the G&S of the study. A social matter (data, performance, impact, stakeholder) is significant (or material) if it is of such relevance and importance that it could substantially influence the conclusions of the study, and the decisions and actions based on those conclusions. Materiality is thus independent from the level of influence that an organization plays on the different phases of the product system under study.

Conclusions, limitations, and recommendations

Conclusions have to be drawn and recommendations made, based on the G&S of the study. It may be best to start with preliminary conclusions and verify if they are consistent with the requirements set out for the study. If these are not consistent, it may be necessary to return to previous steps to address the inconsistencies. If the preliminary conclusions are consistent, then the reporting of the results may proceed. The reporting should be fully transparent, implying that all assumptions, rationales, and choices are identified.

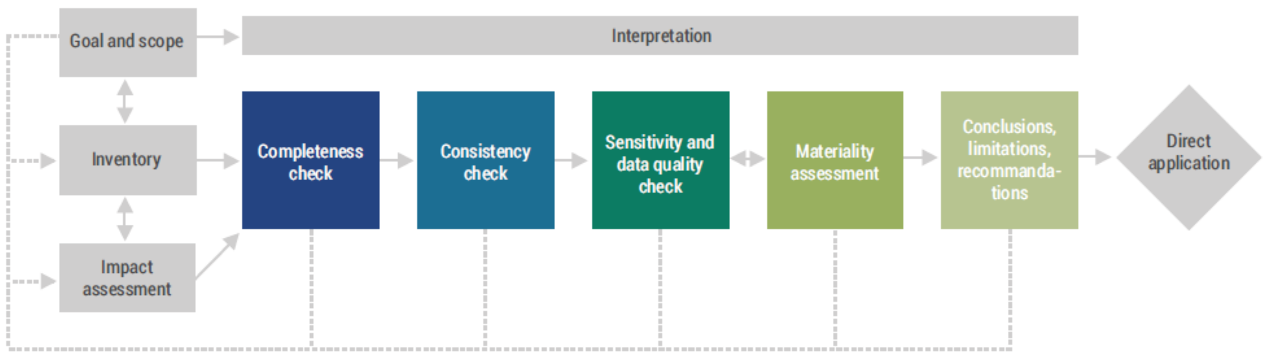


Figure 3. Elements in the interpretation phase of an S-LCA (UNEP, 2020)

S-LCA methodology in PATHWAYS

The S-LCA methodology described in the previous section 2 will be applied in the PATHWAYS project, that has a focus on the livestock sector. This chapter outlines the general ideas for applying the S-LCA methodology in PATHWAYS. Since the specific case studies are not yet known, the details have to be decided upon later. In addition to that, S-LCA is an iterative process, therefore the impacts might be captured in a subsequent iteration of the study. Boundary setting is often performed in an iterative way and the assessment can be improved over time, going from more generic results to more site- and case-specific ones. Revisions might be due to unforeseen limitations or constraints, or due to new additional information.
The first section in this chapter describes the general idea of applying the RS S-LCIA approach in PATHWAYS, since the aim is to describe a product system with a focus on its social performance or social risk. Because the RS S-LCIA is more operational and can be performed for all 40 impact subcategories, this will allow for a broader scope of the study. Nevertheless, it might be the case that the application of the IP S-LCIA approach can be explored in this study, however this can only be done for impact subcategories related to working conditions and quantified impact of farming activities (i.e. human health) (Chen & Holden, 2017) . The applicability of both impact assessment methods to the PATHWAYS project has to be studied in more detail to conclude upon this.
The second section describes the first attempt to select relevant stakeholder categories and impact subcategories for the PATHWAYS project via a combined top-down and bottom-up approach.
In the third section, a first attempt for primary data collection was done specifically at the Practice Hubs, being the stakeholder group workers in the primary production stage of livestock production (i.e., farmers). These initial efforts will help to identify opportunities and challenges in capturing the social impacts in the European livestock sector for the PATHWAYS project.
The fourth section reflects shortly on possible secondary data sources which could be used.

GENERAL IDEA OF APPLYING S-LCA IN PATHWAYS

IMPACT ASSESSMENT WITH REFERENCE SCALE APPROACH

Predominantly, the S-LCIA phase in the PATHWAYS project is conducted according to the RS S-LCIA. This approach enables the assessment of all stakeholder groups and their related impact categories, which makes them compatible with the multi-actor perspective (UNEP, 2020). Moreover, the main S-LCA databases are in line with RS S-LCIA. Figure 4 shows the steps related to the RS S-LCIA approach.

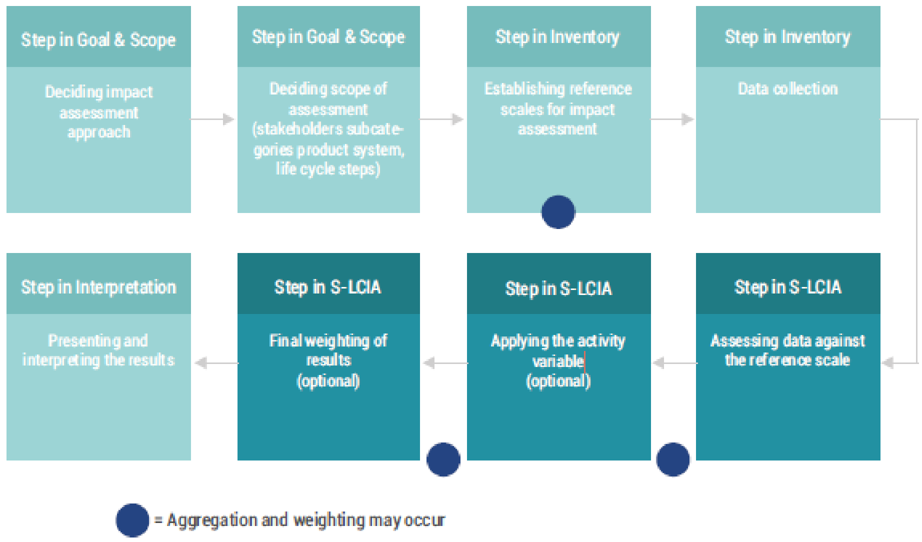


Figure 4. Steps related to the impact assessment process for the Reference Scale approach (UNEP, 2020)

S-LCA generic databases provide data about social risks at country and sector levels. Besides the methodological sheets, the Handbook for Product Social Impact Assessment (PSIA) (Goedkoop et al., 2020) is an important basis for RS S-LCA approaches. The RS S-LCIA phase covers the definition of reference scales that are used to evaluate each of the social inventory performance indicators considered for the product system. Performance Reference Points (PRP) are also determined to allow estimating social risk or performance levels comparing to international standards, local legislations, or organizations' best practices (UNEP, 2020). In the case of generic database use, reference scale and PRP are provided for each social inventory performance indicator. According to the framework defined in this work, social and socio-economic impact (sub)categories that are perceived as the most relevant following the prioritization are used to perform the RS S-LCIA phase. Social inventory performance indicators, performance scales, and PRP are attributed to the selected social and socio-economic subcategories. The calculation is performed following the characterization method chosen for the study.

Reference scales are established during the Inventory phase. It is a crucial preparatory step for organizing inventory data collection and for the implementation of the impact assessment. Reference scales are ordinal scales, typically comprised of 1 to 5 levels, each of which corresponds to a PRP. PRPs are thresholds, targets, or objectives that set different levels of social performance or social risk, which allow to estimate the magnitude and significance of the potential social impacts associated with organizations in the product system. The PRPs are context-dependent and are often based on international standards, local legislation, or industry best practices – normative reference points – or upon other points of reference. Comparing relevant inventory performance indicator data with these levels allows qualifying whether the data collected suggests a negative or a positive performance (of varying degrees in between the two poles).

Reference scales can be ascending – ranging for example from negative performance to positive performance, but they can also be descending – ranging from very low risk to very high risk of potential negative impacts) (UNEP, 2020). They may or may not cover both negative and positive impacts. Reference scales may use numbers to identify the levels or just colours, as depicted in Figure 5.

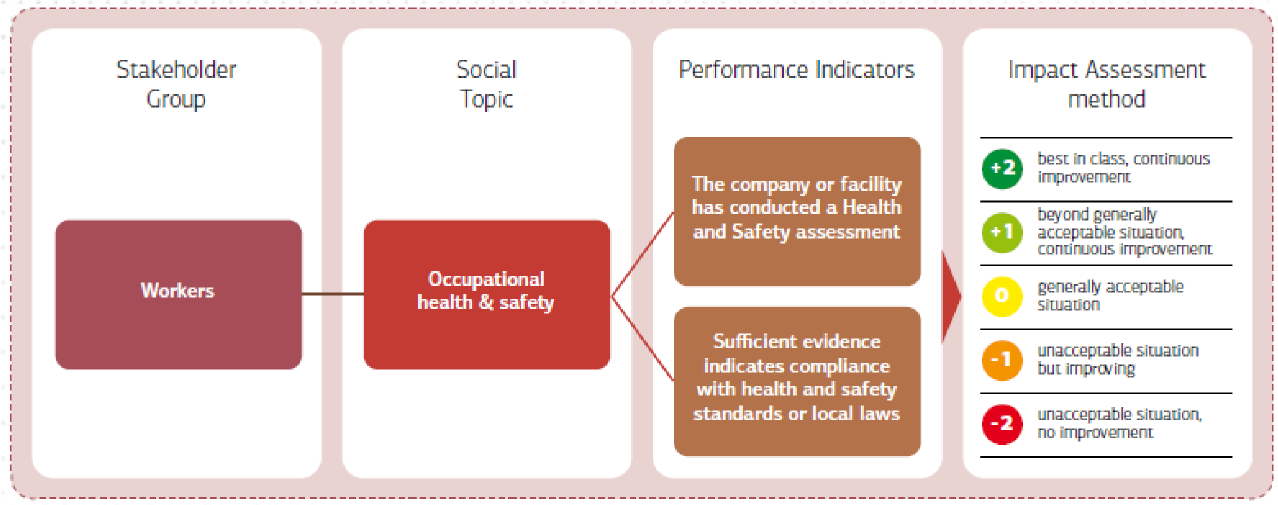


Figure 5. Example of the link between stakeholder group, impact (sub)category and performance indicators for the RS S-LCIA (Goedkoop et al., 2020)

It is recommended not to aggregate positive and negative impacts because impacts can occur on the level of individuals or groups of individuals and, thus, positive impacts might not compensate for negative ones (UNEP, 2020). Presenting the results side by side is acceptable. If, however, aggregated results are needed, the positive and negative impacts shall additionally be shown separately in order to not lose transparency. Aggregation of results should always be done very carefully to avoid misinterpretation and loss of context. This also applies for aggregating results of stakeholder groups, because the location dependent aspect of the results is important – especially when the supply chain is global (UNEP, 2020).

Weights represent the assignment of the relative importance (or contribution) of each performance indicator to the performance of a specific impact (sub)category (UNEP, 2020). During the weighting step, the practitioner applies weights (values) to the inventory, impact (sub)category, or stakeholder category results, to reflect their relative importance (UNEP, 2020). The most common approaches are equal weighting, most robust performance indicators prioritized, export or stakeholder values, and worse performance prioritized (UNEP, 2020).

IMPACT ASSESSMENT WITH IMPACT PATHWAY APPROACH

Impact Pathway Assessment, also known as IP S-LCIA, evaluates the outcomes stemming from the product system, including potential social impacts. This assessment employs one or more characterization models that utilize cause-effect relationships to evaluate impact categories, like E-LCA (Bouillass et al. (2021). The application of IP approach in S-LCA studies differs a lot. In most cases IP approach is applied when it is possible to quantify the impact of a production process with cause-effect chain.

The current development of characterization models within the IP S-LCIA is limited to potential social and socio-economic impacts for a single stakeholder category, mostly the workers, and for a very restricted number of impact (sub)categories (UNEP, 2020). Therefore, this IP S-LCIA is not suited for application in the PATHWAYS project. However, in literature studies assessing S-LCA of the livestock systems it is seen that some studies apply a mixed method including both RS-S-LCIA and IP Pathway approach. For example, the emissions stemming from farm activities can be linked to human health, as is done by Chen & Holden (2017). To understand the viability of this approach we need to first evaluate and test it. Thus, we might explore this IP approach further aligning with the E-LCA on the impact assessment results on emissions and possibility to translate this input into a human health indicator score.

SELECTION OF STAKEHOLDER CATEGORIES AND IMPACT SUBCATEGORIES IN PATHWAYS

For the selection of the stakeholder categories and impact subcategories in PATHWAYS, a combined top-down and bottom-up approach was used to develop a defensible suite of performance indicators, which integrated internationally recognized impact subcategories and the perspectives of affected stakeholders (Kruse et al., 2009a).

In the PATHWAYS project several stakeholders are involved in different forms. National practice hubs and a European multi-actor platform will allow for an engaged co-design of transition pathways whilst innovative living labs will allow for the testing and sharing of innovative solutions. A community of practice will extend the multi-actor approach to a broad range of stakeholders. For a first selection of relevant impact subcategories, all livestock product life cycle stages and all stakeholder categories were taken into account. Subsequently the most relevant ones were selected via a combined top-down and bottom-up approach, as will be elaborated in the following paragraphs.

TOP-DOWN IMPACT SUBCATEGORY AND STAKEHOLDER CATEGORY SELECTION

The top-down approach was used first to select impact subcategories and stakeholder categories that are representative of broadly recognized societal values. (Kruse et al., 2009b; UNEP Life Cycle Initiative & Social

LC Alliance, 2020).Furthermore, stakeholder categories and impact subcategories applied in PATHWAYS need to be relevant for stakeholders within the scope of PATHWAYS . Based on the expert judgement, some of the 40 impact subcategories were deselected because they were less relevant for the scope of the study, i.e., the European livestock sector. This top-down selection resulted in 27 possibly relevant impact subcategories for the livestock sector in Europe (Table 4).

Table 4. List of impact (sub)categories by stakeholder relevant for the PATHWAYS project (in black) as determined by a top-down selection process. The red impact subcategories were deemed insufficiently relevant for the European livestock sector.

Stakeholder category					
Worker	Local community	Value chain actors (excluding consumers)	Consumer	Society	Children
1. Freedom of association and collective bargaining	12. Access to material resources	21. Fair competition	26. Health and safety	31. Public commitments to sustainability issues	38. Education provided in the local communities
2. Child labour	13. Access to immaterial resources	22. Promoting social responsibility	27. Feedback mechanism	32. Contribution to economic development	39. Health issues for children as consumer
3. Fair salary	14. Delocalization and migration	23. Supplier relationships	28. Consumer privacy	33. Prevention and mitigation of armed conflicts	40. Children concerns regarding marketing practices
4. Working hours	15. Cultural heritage	24. Respect of intellectual property rights	29. Transparency	34. Technology development	

5. Forced labour	16.Safe and healthy living conditions	25. Wealth distribution	30. End-of-life responsibility	35. Corruption	
6. Equal opportunities / discrimination	17. Respect of indigenous rights			36. Ethical treatment of animals	
7. Health and safety	18. Community engagement			37. Poverty alleviation	
8. Social benefits / social security	19. Local employment				
9. Employment relationship	20. Secure living conditions				
10. Sexual harassment					
11. Smallholders including farmers					

BOTTOM-UP IMPACT SUBCATEGORY SELECTION

Key stakeholders from the practice hubs in PATHWAYS were consulted in a bottom-up approach to identify key impact subcategories from the 27 impact subcategories identified as potentially relevant in the top-down approach described in previous section. This is the actors' consultation process for the prioritization of relevant impact (sub)categories.

A questionnaire (Appendix 1, page 279) was developed in which the stakeholders could rank the top-down selected impact-subcategories (Table 4) based on the importance from their point of view on a scale from 1 (extremely not important) to 7 (extremely important). The aim of this bottom-up approach was to end up with a list of the 5-6 most relevant impact (sub)categories according to the stakeholders to be consulted in the PATHWAYS project. The questionnaire was published in 10 languages and spread via the PATHWAYS channels and received 77 responses. Respondents included project participants and external stakeholders. Responses were received from stakeholders and covered all assessed countries in Europe as well as all livestock value chains. Because significance of the social impact (sub)categories could differ between stakeholders due to their perspective, the distinctive characteristics of the respondents, such as life cycle stage and geographical area, are also considered as recommend by Bouillass et al. (2021). For example, the differences between the results from respondents in the Netherlands and in Italy were analysed, just as from different stakeholder groups, such as farmers and policy makers.

Figure 6 depicts the main results of the survey, it shows how the impact subcategories selected by means of the top-down approach were ranked on a scale from 1 to 7. The higher the mean, the more relevant the stakeholder rated the impact (sub)category for their livestock value chain in their country. The lower the standard deviation, the more consensus among the stakeholders. Both were considered when analysing the results. The impact subcategories with the highest mean (coloured green) also have a low standard deviation, indicating that the respondents agreed upon a high importance for those impact subcategories. The results from the different groups from the different life cycle stages and geographical areas were quite in-line overall.

The 6 impact subcategories with the highest mean were selected for the S-LCA, being: "Ethical treatment of animals", "Health and safety of workers", "Safe and healthy living conditions", "Health and safety of consumers", "Fair competition" and "Local employment".

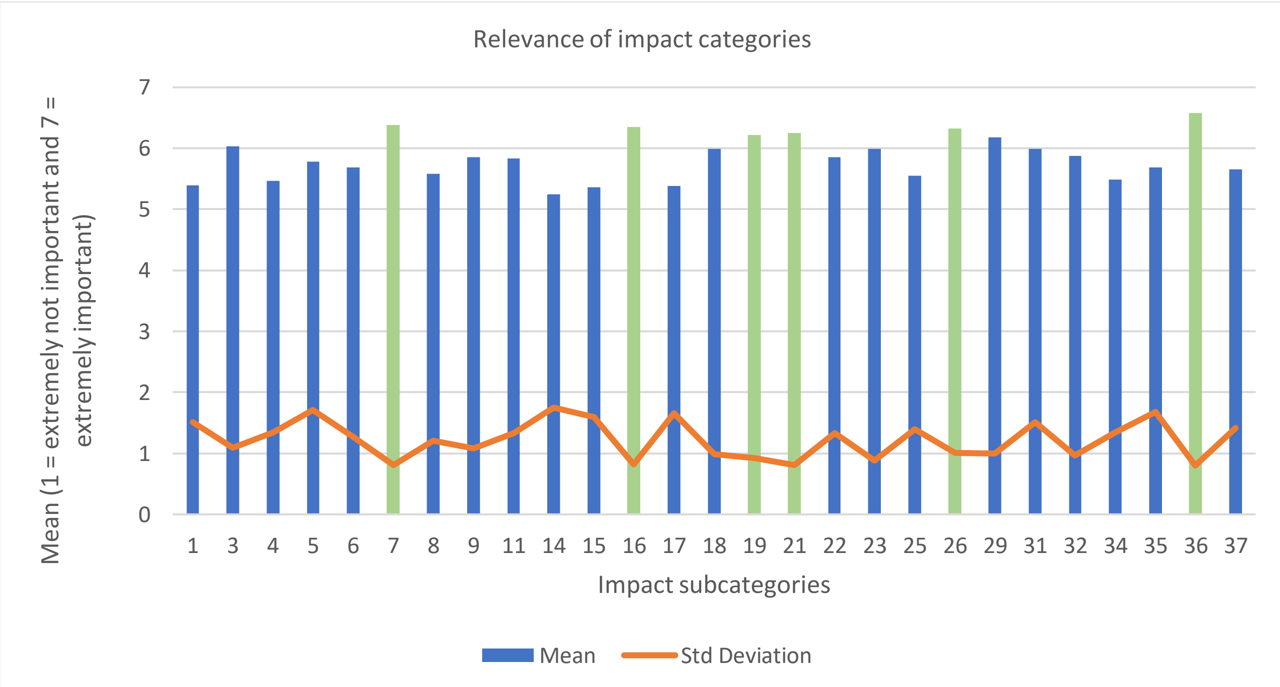


Figure 6. Results of the survey to select most relevant social impact subcategories (numbers refer to the impact category number provided in Table 4). The impact subcategories in green are selected for further analysis in the Pathways project.

PRIMARY DATA COLLECTION FROM PRACTICE HUBS VIA PG-TOOL

In the PATHWAYS project there are Practice Hubs, being groups of innovative farmers in Europe. These Practice Hubs are active in the primary production life cycle stage (or animal farm). Thus, the Practice Hubs represent one part of the whole life cycle of livestock production systems, as depicted in Table 5. Within the PATHWAYS project, data is collected at these Practice Hubs about their sustainability performance with the Public Goods Tool (PG-tool). Researchers visited the participating Practice Hub farmers and assisted the farmers when filling in the PG-tool. We expanded the PG-tool with questions about social impact performance indicators for the following 6 selected impact subcategories "Ethical treatment of animals", "Health and safety of workers", "Safe and healthy living conditions", "Health and safety of consumers", "Fair competition" and "Local employment". Since the PG-tool aims at collecting data from the Practice Hubs, we focus our assessment on the 'Animal farm' life cycle stage. The performance indicators were developed for the Practice Hubs farmers. To be able to assess the entire life cycle, the list of impact subcategories and performance indicators needs to be completed for other stakeholders as well, and these could differ between life cycle stages.

Table 5. Life cycle stages of livestock production

Feed/other inputs	Animal farm	Slaughter and processing	Food manufacturing	Distribution	Retail

The following paragraphs will outline the data collection from the Practice Hubs using the PG-tool, an Excel based whole-farm sustainability assessment. Table 6 depicts the relevant stakeholder categories and impact subcategories identified. Because the PG-tool is a very extensive tool, which takes a substantial amount of time for the farmers to complete, it was decided to include only the most relevant impact subcategories and performance indicators. The impact subcategory "Health and safety of consumers" (i.e. consumer complaints, measures to assess consumer health and safety) was not relevant enough for primary data collection via the PG-tool since it focusses on the 'Animal farm' life cycle stage and was therefore excluded.

Table 6. Relevant stakeholder category and impact subcategories for the life cycle stage 'Animal farm'

Stakeholder	Worker	Local community		Value chain actors	Society
Impact subcategory	Health and safety	Safe and healthy living conditions	Local employment	Fair competition	Ethical treatment of animals

PERFORMANCE INDICATORS FOR THE PG-TOOL

Performance indicators need to be selected for each position on the reference scales and will be used as a guiding principle. The performance indicators are qualitative markers of performance for each of the impact subcategories. Performance indicators do differ per life cycle stage and per stakeholder category and can be formed in an iterative way. The performance indicators can be measured by means of primary and/or secondary data. Therefore, the performance indicators in Table 7 are created specifically for use in the PG-tool for the life cycle stage "Animal farm".

Table 7. Stakeholders, impact subcategories and performance indicators used for the 'Animal farm' life cycle stage in PATHWAYS

Stakeholder	Impact (sub)category	Performance indicators
Worker	Health and safety	Number of injuries when working in your company during the years 2017-2021?
		Number of full time workers, including both employees, contracted, own labour, both paid and unpaid (see rows 24 and 40 in sheet Economic data)
		Number of injured persons per full time worker
		Which of the following actions do you take with regards to the health and safety of your workers?
		- Perform a risk assessment to identify high-risk areas for health and safety
		- Train workers on health and safety procedures
		- Implement a verifiable worker health and safety plan
		- Put in place a worker health and safety performance monitoring system
		- Audited in the last five years on worker health and safety issues
		To which of the following hazards do you take action? (Safety And Health At Work, 2022)
		- Biological hazards (e.g. awareness of the spread of animal diseases, contamination of food and water supplies)
		- Chemical hazards (e.g. awareness of hazards associated with materials such as chemicals, pesticides, herbicides, insecticides, manure, grain storage)
		- Ergonomic hazards (e.g. prevent overuse injuries, learn safe lifting, manual material handling practices)

		- Physical hazards (e.g. use of hearing protection, mechanical ventilation for air contamination)	
		- Psychological hazards (e.g. mental health, fatigue)	
Local community	Safe and healthy living conditions	What proportion of your animals are not accepted at the abattoir inspection due to pathologies/lesions/drug residues (taken from the sheet Animal welfare)?	
		Which efforts do you make to minimize the use of hazardous substances in water (taken from the sheets Water management and Agri-environmental Management)?	
		Which efforts do you make to minimize the use of hazardous substances in soil (taken from the sheet Soil management)?	
		Which efforts do you make to minimize the use of hazardous substances from manure (taken from the sheet Manure and fertiliser)?	
	Local employment	Of your total workforce during the years 2017-2021, what proportion was short-term employed or hired?	
		Of your short-term workforce during the years 2017-2021, what proportion was from the same municipality as your farm?	
		Of your long-term workforce during the years 2017-2021, what proportion was from the same municipality as your farm?	
		Of your total workforce during the years 2017-2021, what proportion was from your family?	
		Do you have policies on local employing/hiring preferences (like people from your municipality)?	
		What percentage of your spending on goods and services during the years 2017-2021 was spent on locally-based suppliers (within your own municipality)?	
Value chain actors	Fair competition	Do you feel like you receive a fair price for your product (taken from the sheet Farm business resilience)?	
		Do you feel like the price you receive covers the costs you make for the product (taken from the sheet Farm business resilience)?	
		Do you experience unfair trade practices (e.g. short-notice cancellations, unilateral contract changes)?	
		Are you satisfied with the trade relationship with your customer(s) (e.g. timely and clear communication)?	
		Is it clear to you how your sales price is determined?	
		Do you feel like risks, costs and profit are fairly divided between you and your buyer(s)?	
Consumer	Health and safety	*Decided to not include this in the primary data collection via the PG-tool.*	
Society		Staff resources	How often are ordinary healthy livestock inspected?

	Ethical treatment of animals		How many times per day are animals at welfare risks (parturition etc.) inspected for signs of illness/injury?
			Are your stock-people regularly trained in relevance of animal welfare?
			Has someone at the farm skills on how to put down e.g. a sick or injured animal?
			Do you have routines for claw trimming?
			How are feed rations for livestock derived?
			Do you have plans and necessary equipment to handle crises, e.g. fire, high temperatures, water shortage or power break down?
		Animal health	How are you working with animal health?
			Do you cooperate with some external, such as a veterinarian or advisor in preventive animal health?
			If you have dairy cows or sheep, how high is the somatic cell counts in delivered milk during the specific year?
			What was the mortality rate in your growing animals during the years 2017-2021? Please include animals dead at arrival at the slaughterhouse but not new-born animals dead within 24 h from birth. Please, give % here Please, give time span here in number of days for the % given above
			What was the mortality rate in your adult animals during the years 2017-2021? Please include animals dead at arrival at the slaughterhouse but not culled animals. Please, give % here Please, give time span here in number of days for the % given above

			What proportion of your animals are not accepted at the abattoir inspection due to pathologies/lesions/drug residues?
			Animals dead at farm, have they been actively put down or passed away by themselves?
		Behaviour	Do your animals graze?
			Do housed animals have access to outdoor areas (taken from Manure and fertiliser sheet)?
			Do you provide any environmental enrichment (pecking material etc.)?
			If you have pigs, are your sows fixated around parturition?
			Is cannibalism and/or injurious behaviour (e.g. feather pecking) occurring?
		Housing	Do housed animals have access to straw or other litter?
			Do housed animals have access to solid floor?
			For growing animals, if any, how much space is available per animal in the most dense group? Please, give average liveweight per animal here, kg Please, give number of animals in the group here Please, give available area here, m2
			For adult animals, if any, how much space is available per animal in the most dense group? Please, give average liveweight per animal here, kg Please, give number of animals in the group here Please, give available area here, m2
		Biosecurity	Are new animals entering the farm kept in a separate barn/section?
			What system for entrance of new animals is practised?
			To what extent do you provide public access to your animals (taken from the sheet System security and diversity)?

SECONDARY DATA COLLECTION

Next to primary data also secondary data might be utilized in the social impact assessment within PATHWAYS. Therefore, secondary data sources which include data for the selected impact categories, "Ethical treatment of animals", "Health and safety of workers", "Safe and healthy living conditions", "Health and safety of consumers", "Fair competition" and "Local employment", need to be utilized. Data on prevalent farming livestock systems in Europe and relevant data on farm size, input, and output, sourced from FADN, FarmDyn, and regional expert input are assessed. Nonetheless, this data is likely insufficient and not entirely relevant for conducting a full social impact assessment solely on secondary data. Hence, secondary data sources might also include EUROSTAT, ILOSTAT, websites of national livestock institutions, literature, and other relevant statistical data sources throughout Europe.

References

Bouillass, G., Blanc, I., & Perez-Lopez, P. (2021). Step-by-step social life cycle assessment framework: a participatory approach for the identification and prioritization of impact subcategories applied to mobility scenarios. International Journal of Life Cycle Assessment, 26(12), 2408–2435. https://doi.org/10.1007/s11367-021-01988-w

Busch, G. (2023). Social aspects of livestock farming around the globe. In Animal Frontiers (Vol. 13, Issue 1). https://doi.org/10.1093/af/vfac084

Chen, W., & Holden, N. M. (2017). Social life cycle assessment of average Irish dairy farm. International Journal of Life Cycle Assessment, 22(9). https://doi.org/10.1007/s11367-016-1250-2

Directorate - General for Agriculture and Rural Development. (2020, October 14). Commission publishes external study on future of EU livestock.

Goedkoop, M. J., de Beer, I. M., Harmens, R., Peter Saling, Dave Morris, Alexandra Florea, Anne Laure Hettinger, Diana Indrane, Diana Visser, Ana Morao, Elizabeth Musoke-Flores, Carmen Alvarado, Ipshita Rawat, Urs Schenker, Megann Head, Massimo Collotta, Thomas Andro, Jean-François Viot, & Alain Whatelet. (2020). Product Social Impact Assessment methodology. www.product-social-impact-assessment.com

Kruse, S. A., Flysjö, A., Kasperczyk, N., & Scholz, A. J. (2009a). Socioeconomic indicators as a complement to life cycle assessment - An application to salmon production systems. International Journal of Life Cycle Assessment, 14(1), 8–18. https://doi.org/10.1007/s11367-008-0040-x

Mancini, L., Valente, A., Barbero Vignola, G., Sanyé Mengual, E., & Sala, S. (2022). Social footprint of European food production and consumption. Sustainable Production and Consumption. https://doi.org/10.1016/J.SPC.2022.11.005

UNEP;, Benoît Norris, C., Traverso, M., Neugebauer, S., Ekener, E., Schaubroeck, T., Russo Garrido, S., Berger, M., Valdivia, S., Lehmann, A., Finkbeiner, M., & Arcese, G. (eds.). (2020). Guidelines for Social Life Cycle Assessment of Products and Organizations 2020. UNEP. http://www.unep.fr/shared/publications/pdf/DTIx1164xPA-guidelines_sLCA.pdf

UNEP Life Cycle Initiative, & Social LC Alliance. (2020). Guidelines for Social Life Cycle Assessment of Products and Organizations 2020. In United Nations Environment Programme (unepe) (Issue 2).

ZIRA, S. , R. E. , I. E. et al. (2020). Social Life Cycle Assessment of Swedish organic and conventional pork production. International Journal Life Cycle Assessment, 25, 1957–1975. https://doi.org/https://doi.org/10.1007/s11367-020-01811-y

Appendix 1: Questionnaire used in the bottom-up impact subcategory selection

S-LCA PATHWAYS

SURVEY ON SOCIAL LIFE CYCLE ASSESSMENT OF LIVESTOCK FOOD SYSTEMS

This survey is conducted within the framework of the PATHWAYS project with the aim to find out which social issues play a role in which livestock value chains.

It will take about 5-10 minutes to fill in this survey.

- ◯ I hereby acknowledge my answers will remain anonymous and for use within the PATHWAYS project and are in accordance with the European Regulation 2016/679.

Are you part of PATHWAYS?

- ☐ Yes, I am a practice hub member
- ☐ Yes, I am a facilitator
- ☐ Yes, I am a multi-actor platform member
- ☐ Yes, I am a community of practice member
- ☐ No, I am external to the project
- ☐ Other ____

Which stakeholder group do you (most) belong to for this project?

- ◯ Farmer
- ◯ Policy maker
- ◯ Citizen / Consumer
- ◯ Research / Academia / Innovation organisations
- ◯ Business / Company
- ◯ Association / Organisation
- ◯ Farm advisory / Veterinarian
- ◯ Food retailer
- ◯ Other ____

In which livestock value chain are you active?

- ☐ Dairy
- ☐ Pork
- ☐ Beef
- ☐ Poultry - meat
- ☐ Poultry - eggs
- ☐ Sheep
- ☐ Goats
- ☐ I am not involved directly with livestock
- ☐ Other ____

In which country do you operate?

- ☐ The Netherlands
- ☐ France
- ☐ Germany
- ☐ Sweden
- ☐ Italy
- ☐ Romania
- ☐ Spain
- ☐ Denmark
- ☐ Poland
- ☐ United Kingdom
- ☐ Switzerland
- ☐ Belgium
- ☐ Other ____

Display This Question:

If Are you part of PATHWAYS? = Yes, I am a facilitator

Or Are you part of PATHWAYS? = Yes, I am a practice hub member

Which of the following Practice Hubs are you a part of?

- [ ] Dairy 1 - FiBL
- [ ] Dairy 2 - ACTA
- [ ] Dairy 3 - USAMVCN
- [ ] Dairy 4 - SLU
- [ ] Pork 5 - ACTA
- [ ] Pork 6 - AU
- [ ] Pork 7 - SEGES
- [ ] Pork 8 - WR
- [ ] Beef 9 - RAU
- [ ] Beef 10 - UNIPI
- [ ] Beef 11 - NATUR
- [ ] Poultry - meat 12 - IUNG
- [ ] Poultry - meat 13 - ACTA
- [ ] Poultry - eggs 14 - AERES
- [ ] Sheep and goats 15 - CSIC
- [ ] Sheep and goats 16 - CSIC

Display This Question:

If Which stakeholder group do you (most) belong to for this project? = Business / Company

What type of company do you work for?

☐ Feed company

☐ Animal trade company

☐ Slaughterhouse / Meat processor

☐ Processing company

☐ Other ____

Display This Question:

If Which stakeholder group do you (most) belong to for this project? = Association / Organisation

Which type of association/organisation do you work for?

☐ Animal welfare organisation

☐ Other ____

Display This Question:

If Which stakeholder group do you (most) belong to for this project? = Farmer

Or Are you part of PATHWAYS? = Yes, I am a practice hub member

What type of farm do you work on?

◯ Conventional

◯ Organic

◯ Other ____

How would you rate the importance of the impact subcategories below for the livestock value chains you are active in?

If you are not active in a specific livestock value chain, please fill this from your stakeholder group point of view. For definitions on the impact subcategories, please see this reference document.

	Extremely not important	Not important	Slightly not important	Neutral	Slightly important	Important	Extremely important	Don't know
Freedom of association and collective bargaining	◯	◯	◯	◯	◯	◯	◯	◯
Fair salary	◯	◯	◯	◯	◯	◯	◯	◯
Working hours	◯	◯	◯	◯	◯	◯	◯	◯
Forced labor	◯	◯	◯	◯	◯	◯	◯	◯
Equal opportunities/discrimination	◯	◯	◯	◯	◯	◯	◯	◯
Health and safety (workers)	◯	◯	◯	◯	◯	◯	◯	◯
Social benefits/social security	◯	◯	◯	◯	◯	◯	◯	◯
Employment relationship	◯	◯	◯	◯	◯	◯	◯	◯
Smallholders including farmers	◯	◯	◯	◯	◯	◯	◯	◯
Delocalization and migration	◯	◯	◯	◯	◯	◯	◯	◯
Cultural heritage	◯	◯	◯	◯	◯	◯	◯	◯
Safe and healthy living conditions	◯	◯	◯	◯	◯	◯	◯	◯
Respect of indigenous rights	◯	◯	◯	◯	◯	◯	◯	◯

	Extremely not important	Not important	Slightly not important	Neutral	Slightly important	Important	Extremely important	Don't know
Community engagement	○	○	○	○	○	○	○	○
Local employment	○	○	○	○	○	○	○	○
Fair competition	○	○	○	○	○	○	○	○
Promoting corporate social responsibility	○	○	○	○	○	○	○	○
Supplier relationships	○	○	○	○	○	○	○	○
Wealth distribution	○	○	○	○	○	○	○	○
Health and safety (consumers)	○	○	○	○	○	○	○	○
Transparency	○	○	○	○	○	○	○	○
Public commitments to sustainability issues	○	○	○	○	○	○	○	○
Contribution to economic development	○	○	○	○	○	○	○	○
Technology development	○	○	○	○	○	○	○	○
Corruption	○	○	○	○	○	○	○	○
Ethical treatment of animals	○	○	○	○	○	○	○	○
Poverty alleviation	○	○	○	○	○	○	○	○

What is your gender?

◯ Female

◯ Male

◯ Prefer not to say

◯ Prefer to self-describe below ____

What is your age?

____

What is your highest education level?

◯ Primary education

◯ Secondary education

◯ Bachelor's or equivalent

◯ Master's or equivalent

◯ Doctorate or equivalent

◯ Other ____

We thank you for your time taking this survey!

For more information on PATHWAYS or to sign up to our newsletter, please visit our website.
In case you have any questions, please reach out to: annabel.oosterwijk@wur.nl

Do you want us to be able to reach out to you?

- ◯ Yes, for another survey/interview. Please leave your e-mailadress.

  ____

- ◯ No