ENVIRONMENTAL LIFE CYCLE ASSESSMENT AND FOOD SYSTEMS:

The Case Study of Agroforestry

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"Learning how to learn is the most important skill to acquire"

– Robert Greene,
'The Daily Laws', p. 51

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Preface

This PhD thesis, entitled "Environmental Life Cycle Assessment and Food Systems: The Case Study of Agroforestry" was prepared between September 2021 and November 2024 to achieve a PhD degree.

The work was conducted in the Department of Agroecology, AU Viborg - Research Centre Foulum, Aarhus University, and submitted to the Graduate School of Technical Sciences (GSTS) at Aarhus University. This thesis was funded by the MIXED project (Grant agreement N° 862357) under the Horizon Europe Framework Program (HORIZON 2020) for project meetings in Denmark, France, and Portugal, conferences in Peru and Italy, and PhD courses in Denmark, France, Switzerland, Sweden, and Barcelona, among others. Additional funding was provided by Aarhus University for international research exchanges in Austria and Portugal.

Aarhus, Denmark, November 2024

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To all my colleagues at Aarhus University, especially those in the section on 'Agricultural Systems and Sustainability' and the group 'Life cycle assessment of agricultural products', thank you for welcoming me over the past three years and for sharing all the diverse and interesting perspectives from your daily work. I deeply appreciate the guidance, training, time, space, availability, flexibility, patience, efforts, understanding, and technical support and feedback that contributed to the positive completion of this work. An additional thanks to everyone I met during my PhD journey who also contributed positively in various ways, whether through direct collaboration as co-authors, teamwork in courses, carpooling to Foulum, discussions at conferences, support and hospitality during research exchanges in Austria and Portugal, or simply by being a friendly and supportive ear during difficult times. You know who you are!

Summary

Modern food systems exert considerable pressure on the biophysical environment. Agroforestry Systems (AFS), proposed as a promising alternative, often lacks comprehensive assessments addressing multiple environmental factors beyond the farm level. Although Life Cycle Assessment (LCA) can help address these gaps, existing LCAs of AFS have inconsistent results due to varied methodologies, complicating cross-study comparisons and a clear understanding of AFS's environmental performance.

In this context, this PhD thesis sets to explore the net environmental and climate impacts associated with different foods in AFS using a supply-side and product-level perspective and an attributional LCA framework combined with a mixed-method case study approach. The central research question was twofold: what insights can LCA provide when comparing the environmental and climate impacts of different food products from AFS, and how do methodological choices affect the outcomes of the assessment? To answer this question, the research design was structured across three specific papers: a comprehensive and systematic review of global AFS LCAs (Paper I), an in-depth case study of silvopastoral systems in Austria (Paper II), and a Life Cycle Inventory development for AFS in Portugal (Paper III).

This PhD shows mixed outcomes, indicating that the environmental performance of AFS is context-dependent and influenced by management practices and methodological choices. Thus, the findings highlight the need for greater standardization, more LCA studies across different agroforestry configurations, and a more nuanced approach to capturing the positive and negative interactions within AFS, which are only partly reflected in current LCAs. Future research should include data on animal behavior in tree-dense systems and refine LCA methodologies with attention to multifunctionality and carbon sequestration.

The contributions of this PhD are (i) a systematic guide to recent research on AFS and LCA, (ii) an expanded systems-level analysis of AFS that captures multiple environmental interactions, (iii) the testing of diverse modeling approaches applicable to similar systems, and (iv) the development of representative inventory data specific to silvopastoral contexts.

Sammendrag

Moderne fødevaresystemer udøver et betydeligt pres på det biofysiske miljø. Agroforestrysystemer (AFS) er blevet foreslået som et lovende alternativ, men mangler ofte omfattende vurderinger, der adresserer flere miljøfaktorer ud over landbruget selv. Selvom livscyklusvurdering (LCA) kan hjælpe med at udfylde disse huller, giver eksisterende LCAs af AFS inkonsekvente resultater på grund af de forskellige metodiske tilgange, hvilket komplicerer tværgående studier og muligheden for en klar forståelse af AFS's miljømæssige påvirkning. I denne sammenhæng søger denne ph.d.-afhandling at undersøge de samlede miljø- og klimaeffekter forbundet med forskellige fødevarer i AFS ud fra et udbud og produktniveauperspektiv ved anvendelse af en attributionel LCA kombineret med en mixed-method case-studie tilgang. Det centrale forskningsspørgsmål var todelt: hvilke indsigter kan LCA give ved sammenligning af de miljø- og klimaeffekter, som forskellige fødevarer fra AFS medfører, og hvordan påvirker metodiske valg vurderingens resultater? For at besvare dette spørgsmål var forskningen strukturelt designet i tre specifikke artikler: en omfattende og systematisk gennemgang af globale AFS LCAs (Artikel I), et dybdegående casestudie af silvopastorale systemer i Østrig (Artikel II) og en livscyklusinventarudvikling for AFS i Portugal (Artikel III). Denne ph.d. præsenterer blandede resultater, som indikerer, at AFS's miljømæssige påvirkning er kontekstafhængig, påvirket af forvaltningspraksis og metodiske valg. Disse resultater fremhæver derfor behovet for mere standardisering, flere LCA-studier på tværs af forskellige agroforestrykonfigurationer, samt en mere nuanceret tilgang til at indfange de positive og negative interaktioner i AFS, som kun delvist er afspejlet i nuværende LCAs. Fremtidig forskning bør inkludere data om dyreadfærd i træ-tætte systemer og raffinere LCA-metoder med fokus på multifunktionelle systemer og kulstofbinding.

Bidragene fra denne ph.d. omfatter (i) en systematisk vejledning til den nyeste forskning om AFS og LCA, (ii) en bred analyse på systemniveau af AFS, der indfanger flere miljøinteraktioner, (iii) afprøvning af forskellige modeltilgange anvendelige på lignende systemer samt (iv) udviklingen af repræsentative inventardata specifikt for silvopastorale kontekster.

List of publications

Included papers in this thesis

Paper I: **Quevedo-Cascante Monica**, Mogensen Lisbeth, Kongsted Anne Grete, Knudsen Marie Trydeman (2023) 'How does Life Cycle Assessment capture the environmental impacts of agroforestry? A systematic review', *Science of The Total Environment*, 890, 164094. Available at: https://doi.org/10.1016/j.scitotenv.2023.164094, **[Published]**

Paper II: **Quevedo-Cascante Monica**, Dorca-Preda Teodora, Mogensen Lisbeth, Zollitsch Werner, Waqas Muhammad Ahmed, Hörtenhuber Stefan, Geßl Reinhard, Kongsted Anne Grete, Knudsen Marie Trydeman (2024) 'Life Cycle Assessment and modeling approaches in silvopastoral systems: a case study of egg production integrated in an organic apple orchard', *Journal of Environmental Management*, 372, 123377. Available at: https://doi.org/10.1016/j.jenvman.2024.123377 **[Published]**

Paper III: **Quevedo-Cascante Monica**, Dorca-Preda Teodora, Marinheiro Joana, Loureiro João, Mogensen Lisbeth, Knudsen Marie Trydeman (2024) 'Development of a Life Cycle Inventory for a Silvopastoral System in the Montado, Portugal', **[Draft]**

Additional publications

Paper IV: **Quevedo-Cascante, M.,** Mogensen, L., Kongsted, A. G., & Knudsen, M. T. (2024). Data for "How does Life Cycle Assessment capture the environmental impacts of agroforestry? A systematic review" [Data set]. Zenodo. https://doi.org/10.5281/zenodo.10462646. **[Published]**

Paper V: Zhen Huayang, Feng Xu, Waqas Muhammad Ahmed, **Quevedo-Cascante Monica**, Ju Xuehai, Qiao Yuhui, Lohrum Nele, Knudsen Marie Trydeman (2023) 'Solutions to neutralize greenhouse gas emissions of the rice value chain — A case study in China', Sustainable Production and Consumption, 35, pp. 444–452. Available at: https://doi.org/10.1016/J.SPC.2022.11.023. **[Published]**

Paper VI: Goglio Pietro, Knudsen Marie Trydeman, Van Mierlo Klara, Röhrig Nina, Fossey Maxime, Maresca Alberto, Hashemi Fatemeh, Waqas Muhammad Ahmed, Yngvesson Jenny, Nassy Gilles, Broekema Roline, Moakes Simon, Pfeifer Catherine, Borek Robert, Yanez-Ruiz David, **Quevedo-Cascante Monica**, Syp Alina, Zylowsky Tomasz, Romero-Huelva Manuel, Smith Laurence G (2023) 'Defining common criteria for harmonizing life cycle assessments of livestock systems', *Cleaner Production Letters*, 4, p. 100035. Available at: https://doi.org/10.1016/J.CLPL.2023.100035. **[Published]**

Conference paper I:	**Quevedo-Cascante Monica**, Mogensen Lisbeth, Kongsted Anne Grete, Knudsen Marie Trydeman (2022a) 'How does LCA capture the environmental impacts of agroforestry systems? An illustrative case study', in Proceedings of the 13th International Conference on Life Cycle Assessment of Food. Lima, Peru: LCA FOODS, pp. 845–846. **[Published]**
Conference paper II:	**Quevedo-Cascante Monica**, Mogensen Lisbeth, Kongsted Anne Grete, Knudsen Marie Trydeman (2022b) 'The environmental impacts of agroforestry in agri-food systems: a life cycle assessment approach', in Book of abstracts of the 6th European Agroforestry Conference: Agroforestry for the green deal transition. Research and innovation towards the sustainable development of agriculture and forestry. Nuoro, Italy: EURAF, pp. 127–128. **[Published]**
Conference paper III:	Goglio Pietro, Knudsen Marie Trydeman, Van Mierlo Klara, Röhrig Nina, Fossey Maxime, Maresca Alberto, Hashemi Fatemeh, Waqas Muhammad Ahmed, Yngvesson Jenny, Nassy Gilles, Broekema Roline, Moakes Simon, Pfeifer Catherine, Borek Robert, Yanez-Ruiz David, **Quevedo-Cascante Monica**, Syp Alina, Zylowsky Tomasz, Romero-Huelva Manuel, Smith Laurence G (2023) 'Identifying criteria for harmonizing life cycle assessments of crop-livestock systems and interactions'. 52nd Conference of the Italian Society of Agronomy. Royal Palace of Portici, Italy. **[Published]**

Author's contribution

This PhD thesis, comprising **Paper I**, **Paper II**, and **Paper III** is the result of collaborative work led by Mónica Quevedo-Cascante during her PhD fellowship at the Department of Agroecology, Aarhus University, within the framework and constraints set by the MIXED project (funded by the European Union's Horizon 2020, grant number 862357) and Work Package 4. Mónica led the work on all three papers with input from her supervisor, co-supervisors, and co-authors. For **Paper II** and **Paper III**, a survey protocol was collaboratively predeveloped and predesigned in 2021, before the start of the PhD, by participants from the MIXED project (ten national teams) and led by Work Package 2 (Lisbeth Mogensen and Anne Grete Kongsted). The first round of *in situ* data collection using the aforementioned survey protocol (involving up to fourteen farms relevant to this PhD thesis) was conducted by local network partners in November 2022 in Austria (Reinhard Geßl) and June 2023 (João Loureiro and Joana Marinheiro) in Portugal. The second round of *in situ* and *ex situ* data collection and validation was carried out by Mónica during late summer and fall in 2023, in collaboration with local partners in Austria (Werner Zollitsch, Reinhard Geßl, Stefan Hörtenhuber, and Gwendolyn Prehofer) and Portugal (Joana Marinheiro, João Loureiro, João Oliveira, and Carolina Ramos), incorporating farmers feedback when possible (with identities kept confidential).

Furthermore, some introductory text in **Chapter 1** and one illustration (**Figure 2-1**) has been re-used from the qualifying exam progress report submitted and written by Quevedo-Cascante (2023). **Appendix A** and **Appendix** B are identical extracts from **Paper I** published manuscript in the *Science of The Total Environment* and **Paper II** published manuscript in the *Journal of Environmental Management*, respectively. Each paper in **Chapter 3** is a summarized and concise version of the original published and drafted manuscripts (**Paper I** and **Paper II**, and **Paper III**, respectively).

CRediT authorship contribution statement (**Paper I**):

Mónica Quevedo-Cascante: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft. **Lisbeth Mogensen:** Conceptualization, Investigation, Supervision, Validation, Writing – review and editing. **Anne Grete Kongsted:** Writing – review and editing. **Marie Trydeman Knudsen:** Conceptualization, Investigation, Project administration, Supervision, Validation, Visualization, Writing – review and editing.

CRediT authorship statement (**Paper II**):

Mónica Quevedo-Cascante: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft. **Teodora Dorca-Preda:** Conceptualization, Formal analysis, Investigation, Supervision, Methodology, Validation, Writing – review and editing. **Lisbeth Mogensen:** Conceptualization, Formal analysis, Investigation, Supervision, Methodology, Validation, Writing – review and editing. **Werner Zollitsch:** Conceptualization, Investigation, Validation, Writing – review and editing. **Muhammad Ahmed Waqas:** Formal analysis, Investigation, Validation, Writing – review and editing. **Stefan Hörtenhuber:** Investigation, Validation, Writing – review and editing. **Reinhard Geßl:** Conceptualization, Data curation, Investigation. **Anne Grete Kongsted:** Data curation. **Marie Trydeman Knudsen:** Conceptualization, Formal analysis, Investigation, Methodology, Project administration, Supervision, Validation, Writing – review and editing.

CRediT authorship statement (**Paper III**):

Mónica Quevedo-Cascante: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing–original draft.; **Teodora Dorca-Preda:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Writing–review and editing; **Joana Marinheiro:** Data curation, Investigation, Validation, Writing–review and editing; **João Loureiro:** Data curation, Investigation, Validation, Writing–review and editing; **Lisbeth Mogensen:** Conceptualization, Supervision, Methodology, Validation, Writing – review and editing; **Marie Trydeman Knudsen:** Conceptualization, Supervision, Methodology, Validation, Writing–review and editing, Project administration.

Abbreviations

AFS	Agroforestry Systems
ALCA	Attributional Life Cycle Assessment
AP	Acidification Potential
C	Carbon
CF	Carbon Footprint
CH_4	Methane
CO_2	Carbon dioxide
CPS	Crop production systems
C-seq	Carbon sequestration
EP	Eutrophication Potential
F1	Farm 1
F2	Farm 2
FU	Functional unit
FW	Fresh weight
GHG	Greenhouse gas
LCA	Life cycle assessment
LCI	Life Cycle Inventory Analysis
LCIA	Life Cycle Impact Assessment
LO	Land Occupation
LW	Live weight
M1	Model 1
M2	Model 2
M3	Model 3
MIXED	Multi-actor and transdisciplinary development of efficient and resilient MIXED farming and agroforestry systems
MPS	Milk production systems
MTPS	Meat production systems
N	Nitrogen
N_2O	Nitrous oxide
NO_3	Nitrate
NO_X	Nitrogen oxides
RS	Reference systems
RS-A	Reference System for apples
RS-E	Reference System for eggs
SO_2	Sulfur dioxide
UAO	Unit of analysis and observation

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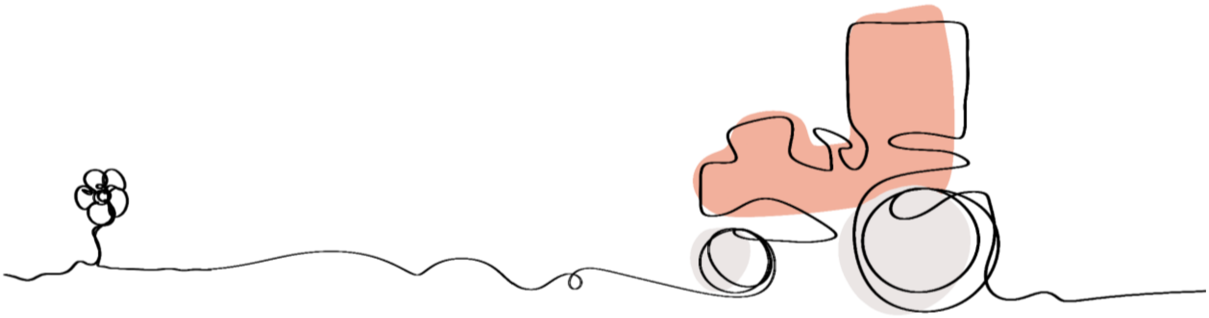


Chapter 1

1. Introduction

This chapter provides the foundational context for this PhD thesis. It begins with an introduction to the global food system and the environmental role of agroforestry as an alternative production pathway. The application of life cycle assessment in the context of agroforestry systems is briefly explored. The problem statement identifies key research gaps and challenges in the general agroforestry literature and life cycle assessment studies. The central research question, along with specific sub-questions and objectives, is then presented to guide the thesis and corresponding papers. The chapter ends by presenting the scientific novelty and relevance of the research.

1.1. Food systems and agroforestry

Behind every meal lies a vast and interconnected global food system essential to nourishing populations and supporting livelihoods worldwide (FAO, 2017b). This system encompasses an interconnected value chain involving a diverse array of stakeholders, including farmers, processors, retailers, and consumers from production to consumption (FAO, 2009). At its core, the journey from farm to table involves a complex matrix of social, economic, and political components. Equally important is the biophysical environment (hereafter referred to as the environment), which includes biotic and abiotic resources crucial for food production and ecosystem functions, such as soil, plants, animals, and energy (Notarnicola *et al.*, 2017).

Today's modern and industrial food system, characterized by high-input monocultures and intensive livestock operations, depends heavily on the continuous supply of inputs such as water, land, fertilizers, and pesticides (Ritchie and Roser, 2022). This approach has led to more intensive and homogenized production systems that exert significant pressure on the environment (**Figure 1-1**), particularly during the production phase in the food value chain (Gliessman, 2014). Food production accounts for 26% of global greenhouse gas (GHG) emissions and it's responsible for 70% of freshwater withdrawal and 78% of ocean and freshwater eutrophication (Ritchie and Roser, 2022). Agricultural emissions, such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), are significant drivers of climate change, contributing about one-quarter of annual GHG emissions (including forestry and land-use change) (IPCC, 2019). However, not all food is produced in the same way and some products are more resource-intensive and responsible for the higher share of impacts. Animal production systems alone account for 14.5% of the total human-induced GHG emissions, mostly linked to CH_4, a gas 25 times the global warming potential of CO_2 (Ahmed *et al.*, 2020). With farmed animals making up 94% of non-human mammal biomass and poultry accounting for 71% of bird biomass, business-as-usual has led to significant environmental degradation worldwide, including soil erosion, water scarcity, deforestation, and biodiversity loss, among others (Notarnicola *et al.*, 2017).

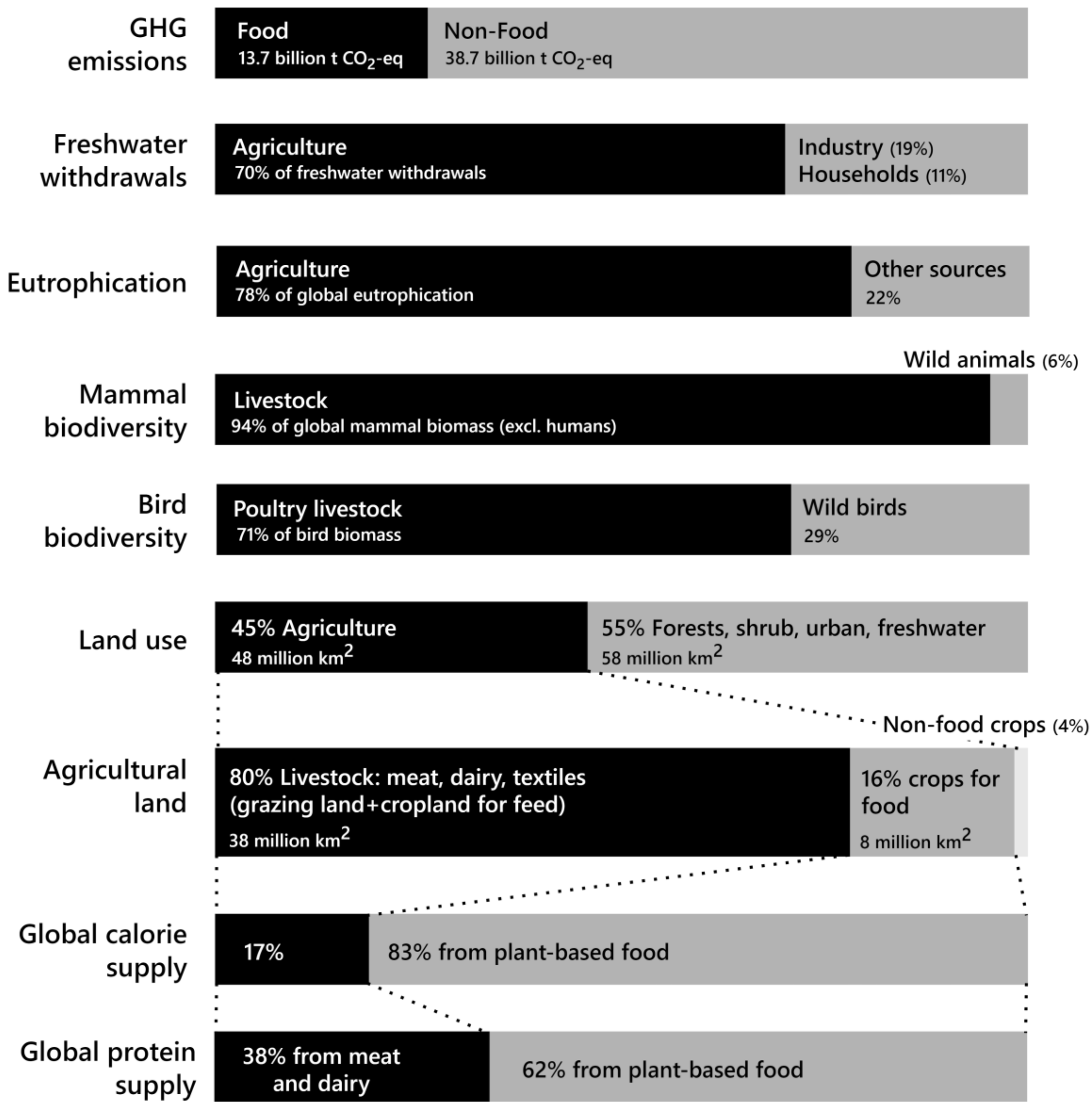


Figure 1-1. Environmental impacts of the agricultural sector, global and agricultural land use, and global calorie and protein supply from different food production systems (figure adapted from: Ritchie and Roser, 2019 and 2022. Data sourced from FAO, 2011; Bar-On *et al.*, 2018; Poore and Nemecek, 2018)

With a growing population, rising economic wealth, unhealthy consumption patterns, concentration of power, food loss and waste, artificial market structures, and unequal distribution of resources (Ahmed *et al.*, 2020), an important concern for the green transition is tackling what humans eat and how it is produced. Innovative approaches are therefore essential for addressing the dual challenges of producing nutritionally adequate food while staying within the planet's environmental boundaries (Rockström *et al.*, 2020). Arguably, the current food production model requires significant transformation, yet the question of how to achieve this remains open and highly disputed. Among the many potential alternatives, some scholars suggest that the adoption of mixed production systems (e.g., the combination of annual or perennial crops with forestry, fisheries, or animals) can reduce pressure on the environment by enhancing biodiversity, improving soil health, and sequestering carbon (Gliessman, 2014). In theory, these systems can be low-input and (partially) closed (e.g., manure is recirculated back as fertilizer for annual crops) (Lantinga *et al.*, 2004). Food from agroforestry is an example of such a potential alternative pathway often discussed in the literature on mixed production systems.

Although agroforestry covers around 43% of the global agricultural land (land with at least 10% tree cover) (Bettles *et al.*, 2021) and is considered a long-standing practice prevalent in rural areas in the Global South (e.g., Asia), it is also a relatively new practice in many regions, having been integrated into national policies only in the 1980s and 2000s (Santiago-Freijanes *et al.*, 2018). In Europe, for instance, agroforestry accounts for approximately 8.8% of the total agricultural area (EU-28), including grazed shrublands and home gardens (den Herder *et al.*, 2017; Burgess and Rosati, 2018). However, the extent is difficult to quantify, partly because a universal definition or common political and quantifiable consensus has yet to be established.

Different concepts and ideas have been suggested to define agroforestry (Atangana *et al.*, 2014a). Broadly, agroforestry is understood as an integrated land-use management system in which woody perennials are deliberately planted sequentially or simultaneously within crop and/or animal systems on the same land (Leakey, 2017) (**Figure 1-2**). Typically, agroforestry systems (AFS) are categorized into four types of configurations according to the nature of their components (**Figure 1-3**). Namely, silvopastoral systems (integrating woody perennials and animals), agrosilvicultural systems (combining crops and woody perennials), agrosilvopastoral systems (mixing crops, woody perennials, and animals), and other systems (mixed of different woody perennials, crops, animals, insects, or aquatic animals) (Atangana *et al.*, 2014a). Each configuration offers distinctive interactions and can result in different environmental outcomes depending on their context (e.g., climate and region) and management approach (e.g., organic or conventional agroforestry) (Köthke *et al.*, 2022).

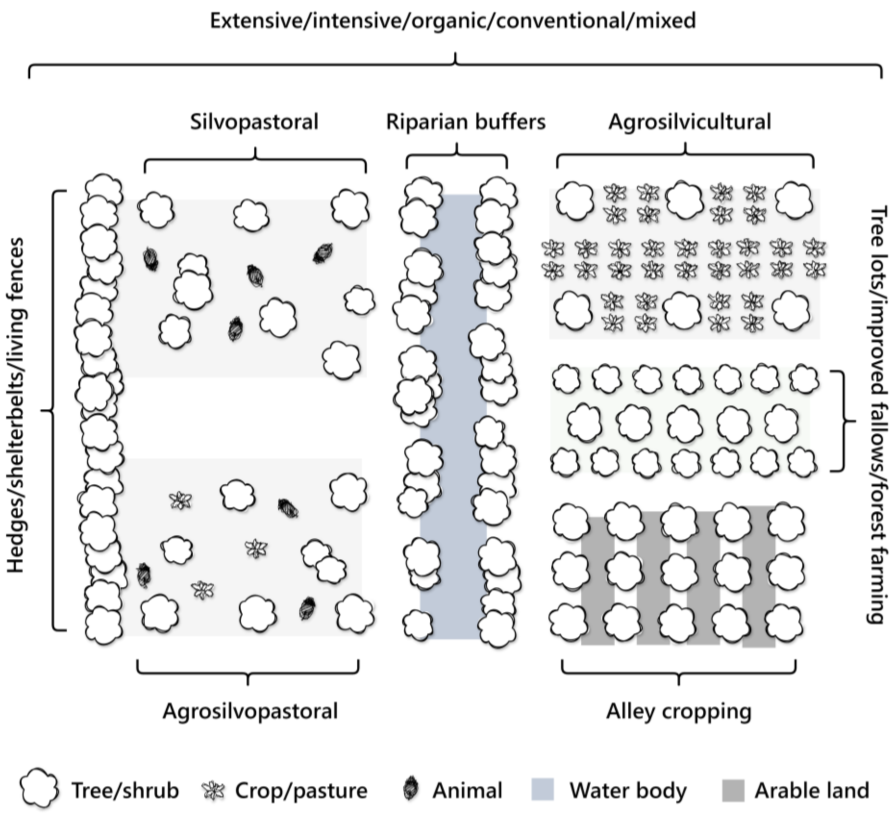


Figure 1-2. Graphical examples of different types of agroforestry configurations and management approaches.

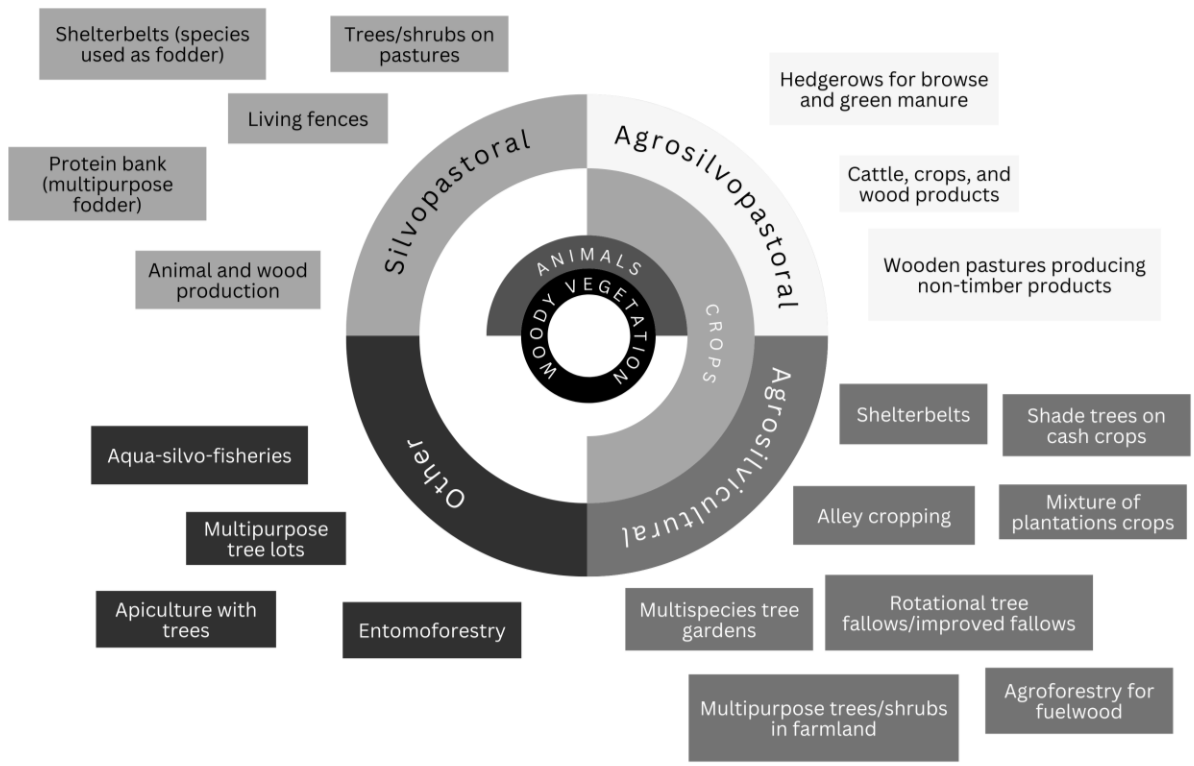


Figure 1-3. Structural composition of agroforestry systems according to their natural components and examples of typical subsystems (information extracted and modified from Nair, 1985 in Atangana *et al.*, 2014).

In theory, AFS are designed to optimize the interactions between the different components in the environmental matrix. In practice, however, the environmental impacts of AFS have been documented in the scientific literature with both, convergent and divergent results (Ollinaho and Kröger, 2021; Köthke *et al.*, 2022). An example is the case of agroforestry eucalypts *(Eucalyptus spp.)*, which has been linked to positive ecological interactions in India (e.g., increased soil organic matter and soil enrichment and fertility in well-drained areas) (Raj *et al.*, 2016; Jose and Udawatta, 2021), but also negative interactions in semi-arid or dry areas in Brazil and Africa, respectively (e.g., water depletion, decline plant biodiversity, and resource competition) (Raj *et al.*, 2016; Borges *et al.*, 2020; Jose and Udawatta, 2021; Wander *et al.*, 2022). These contrasting examples show the importance of considering region-specific aspects, as what works in one area may not in another.

Another example of the varying effects of AFS on the environment can be found in a meta-analysis from Feliciano *et al.* (2018). The authors show that AFS can have different carbon sequestration (C-seq) rates in the soil and the woody biomass, which are more significant in tropical climates and where degraded lands have been converted into improved fallows or silvopastures. The authors also show that sequestration can be significantly lower in temperate, semi-arid, or arid climates. Regardless, the broader agroforestry GHG balance is more nuanced and some scholars argue that it needs to be considered (Alemu, 2014). This is because although AFS can act as carbon sinks, they can also be a source of CH_4 and N_2O emissions, particularly in ruminant-based configurations (Dixon, 1995; Alemu, 2014). In these cases, CH_4 from enteric fermentation and N_2O from manure can offset the C-seq gains, especially in lower latitudes where emissions tend to be higher due to warm temperatures and poorer manure management practices (Dixon, 1995; Alemu, 2014).

A meta-analysis by Torralba *et al.* (2016) shows that European agroforestry can be beneficial to biodiversity (i.e., species richness and abundance), and can have a significant positive effect on the bird taxonomic group. According to the authors, silvopastoral and silvicultural systems provide greater biodiversity than specialized systems or forestry lands (Torralba *et al.*, 2016). However, a more recent meta-analysis suggests that while AFS may provide some biodiversity benefits, the effects are often weak and context-dependent, with no clear overall positive impact on biodiversity across different taxa (Mupepele *et al.*, 2021).

Additionally, in terms of nitrogen (N) dynamics, agroforestry presents both opportunities and challenges. Trees in AFS can enhance N retention in soils by reducing leaching and runoff and promoting mineralization from organic matter inputs for further plant availability (Jose, 2009). As shown in past research, 75-80% of the N leached can be reduced in paddocks with 20% tree cover compared to grass areas (Manevski *et al.*, 2019). However, competition for N between trees, crops, and soil microbes can reduce N availability for crop uptake and can impact yields (Atangana *et al.*, 2014b; Kim and Isaac, 2022). If AFS are intensively or conventionally managed, the N inputs tend to increase, leading to higher N_2O emissions, which can further complicate the GHG balance (Kim and Isaac, 2022). In silvopastoral systems in Brazil, N_2O and CH_4 emissions derived from cattle excreta have been documented ten times higher than in monoculture pasture, especially during the rainy season (Bretas *et al.*, 2020). On the other hand, agrosilvicultural systems (shelterbelts, tree plantations, riparian buffers) have been reported to emit less N_2O emissions compared to conventional crop fields, likely because the N inputs were lower (Kim and Isaac, 2022).

The varying conclusions regarding the effects of AFS on the environment could be partly because different levels of analysis have been used in agroforestry research. This issue is highlighted by Fagerholm *et al.* (2016) in their systematic map of 71 studies in Europe, which found that agroforestry research is often conducted within a single location (70% mainly field measurements) and typically at a small scale (58% of the studies only include one biophysical indicator). Schuler *et al.* (2022) similarly emphasize this limitation in another systematic map of 158 studies of agroforestry and ecosystem services in Brazil. In addition, both studies show different scopes of analysis, where some agroforestry configurations (e.g., wood pastures) have received more attention than others (e.g., riparian buffers).

These limitations in scope and scale are intensified by the diverse range of scientific approaches used to analyze AFS. For example, gas chambers have been utilized to measure GHG emissions from hedgerows, shelterbelts, and silvopastures at the plot level (Kwak *et al.*, 2019). N balance models have been used to calculate nitrate (NO_3)

leaching from outdoor pig production integrated with poplar trees at the paddock level (Manevski *et al.*, 2019). Geospatial technology has been applied to estimate C-seq in small-scale agroforestry farming at the landscape level (Milne *et al.*, 2013). These approaches, while relevant in their respective fields, tend to focus only on one variable (e.g., carbon) and do not include the effects of raw material extraction or other emissions in the value chain. Thus, results at the plot, paddock, or landscape level may contradict those at the systems level, highlighting a need for cross-disciplinary and multi-criteria tools that can comprehensively assess these complex systems, while capturing all the important life cycle stages.

1.2. Life Cycle Assessment and agroforestry

Grasping the intricacies of food systems and applying systems thinking is essential for identifying and prioritizing environmental areas where weaknesses occur (FAO, 2017a). Through systems thinking, it is possible to identify critical issues resulting from interactions between all components within the food value chain and, therefore, develop effective environmental strategies to address them (FAO, 2017a). Life cycle assessment (LCA) is one of the most comprehensive and widely used methods for evaluating the environmental impacts of products and services from a system's perspective (Arvanitoyannis, 2008). LCA is a quantitative tool and a standardized method based on life cycle thinking principles, which analyzes environmental issues (ISO, 2006a). This includes all stages from raw material extraction, manufacturing, and distribution, to use and disposal. LCA provides a comprehensive view of the environmental aspects and potential impacts associated with a product or system, enabling the identification of areas for improvement (Baumann and Tillman, 2004). By considering multiple environmental indicators, LCA helps to avoid shifting burdens from one stage of the life cycle to another or from one environmental medium to another, thus supporting a more holistic analysis.

Modeling biological systems, however, can be a challenging process, let alone multifunctional systems such as agroforestry (Ciroth *et al.*, 2021). While AFS have been analyzed in the LCA literature in the context of free-range poultry and olive orchards (Paolotti *et al.*, 2016), extensive organic livestock systems (Eldesouky *et al.*, 2018; Horrillo *et al.*, 2020), dairy farming (Brook *et al.*, 2022; Ruiz-Llontop *et al.*, 2022), and cocoa production (Utomo *et al.*, 2016; Bianchi *et al.*, 2021), the results often vary (Quevedo-Cascante *et al.*, 2023). This is because AFS consists of multiple interconnected subsystems (e.g., trees, crops, animals) that do not function in isolation. Thus, the interdependent and multifunctional nature of AFS means that small changes in one part of the subsystem (e.g., feed, fertilizers) can lead to significant changes in others (e.g., crop growth, tree health, or animal behavior) (Neven, 2014; FAO, 2017a; Onat *et al.*, 2017). These changes are further amplified by the different methodological choices used in an LCA.

1.3. Problem Statement

> Today's modern food production system significantly impacts the biophysical environment. While agroforestry has been proposed as a promising alternative, current research typically lacks comprehensive assessments that encompass multiple environmental variables and environmental indicators beyond the field or farm level. Although LCA is a potential tool to bridge these knowledge gaps, existing LCAs of AFS have produced varying results due to differences in methodological choices and modeling approaches, making it difficult to compare studies and have a clear understanding of the net environmental impacts of AFS.

1.4. Research questions and objectives

The **central objective** of this PhD thesis is as follows:

> To explore the use of the LCA methodology as a tool for quantifying the net environmental and climate impacts associated with food production in AFS.

The **central research question** of this PhD thesis is as follows:

> What insights can LCA provide when comparing the environmental and climate impacts of different food products from AFS, and how do methodological choices affect the outcomes of the assessment?

This question is unfolded through the following sub-questions:

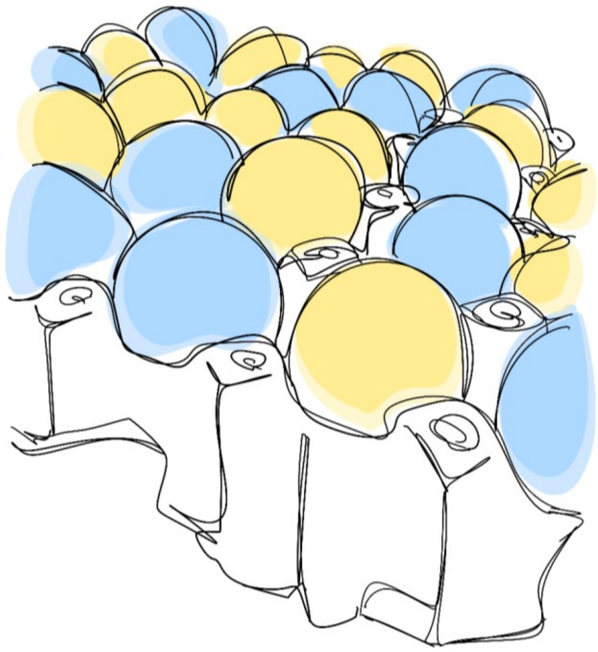
- What is the current state of knowledge on global food LCA studies in the context of AFS?
- What are the environmental and climate impacts of silvopastoral systems and how do different modeling approaches affect the interpretation of LCA results?

The research questions are further elaborated into three specific objectives addressed in three corresponding papers:

Objective 1: To systematically synthesize and explore the scientific evidence on the methodological choices within each phase of the LCA framework and the environmental performance of agri-food products from global AFS (**Paper I**).
Objective 2: To test different modeling approaches and broaden the scope of analysis of food produced in a silvopastoral system in Austria (**Paper II**).
Objective 3: To prepare a Life Cycle Inventory for a silvopastoral system in Portugal (**Paper III**).

1.5. Scientific novelty and relevance

This PhD is relevant because it bridges three interconnected yet often isolated fields – LCA, agroforestry, and food systems. It is novel because it contributes to the scientific literature in four ways. First, it is the first to systematically guide scholars to the latest research on AFS and LCA. Second, it expands the focus of attention of AFS to a systems-level issue while addressing multiple environmental interactions and impact categories. Third, it is the first to test various modeling methods for handling multifunctionality applicable to other systems, and the first to include C-seq potential in apple and egg systems. Finally, it is the first to develop representative inventory data for a natural cork oak silvopastoral plantation.



Chapter 2

2. Methodology

This chapter sets out the methodological and conceptual foundation used in this PhD. It begins by introducing the background of the PhD, including the funding project and the predefined contextual setting and pool of case studies. It then presents the research framework and provides the conceptual structure and overarching methodological approach that guides the PhD. Finally, it describes the research design and refers to the specific strategy applied to answer the research questions, such as the rationale for selecting certain environmental indicators.

2.1. Background

This PhD was part of MIXED ('Multi-actor and transdisciplinary development of efficient and resilient MIXED farming and agroforestry systems'), an international, cross-sectional, transdisciplinary, and multi-actor research project funded by the European Union under the Horizon2020 program. MIXED is conducted in a consortium with 14 farmer networks across 10 European countries. The predesignated networks for this PhD were Austria and Portugal (**Figure 2-1**) and the scientific work was constraint by the predefined tasks in Work Package 4 (i.e., assessing the environmental and net climate impact along the value-chain of products from AFS using attributional LCA).

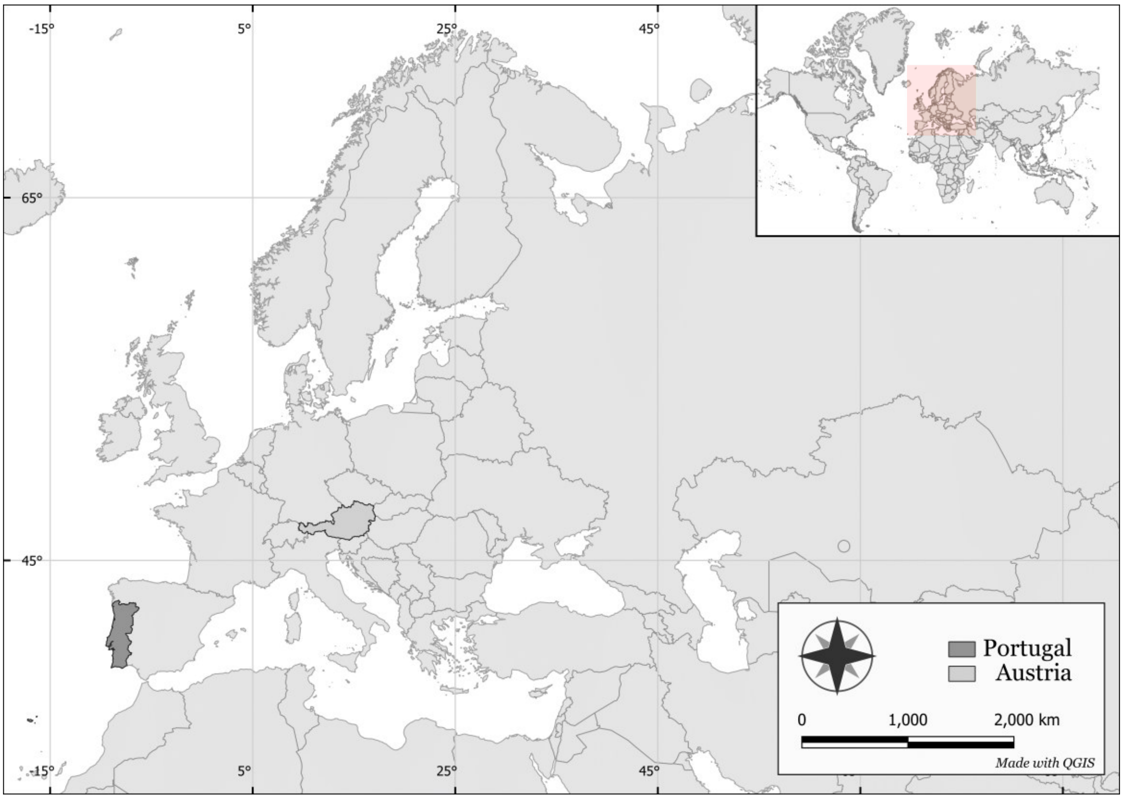


Figure 2-1. Map of study areas (Portugal and Austria) part of the MIXED project and Work Package 4 (Quevedo-Cascante, 2023).

2.2. Research framework

The overall methodological framework in this thesis was underpinned by the Life Cycle Assessment (LCA) framework and guided by an explorative and pragmatic research paradigm, including an iterative process between the four phases of the LCA. LCA is a standardized methodology for quantifying the environmental performance of a product throughout its entire life cycle (Baumann and Tillman, 2004; ISO, 2006b, 2006a). The LCA framework is composed of four phases (**Figure 2-2**).

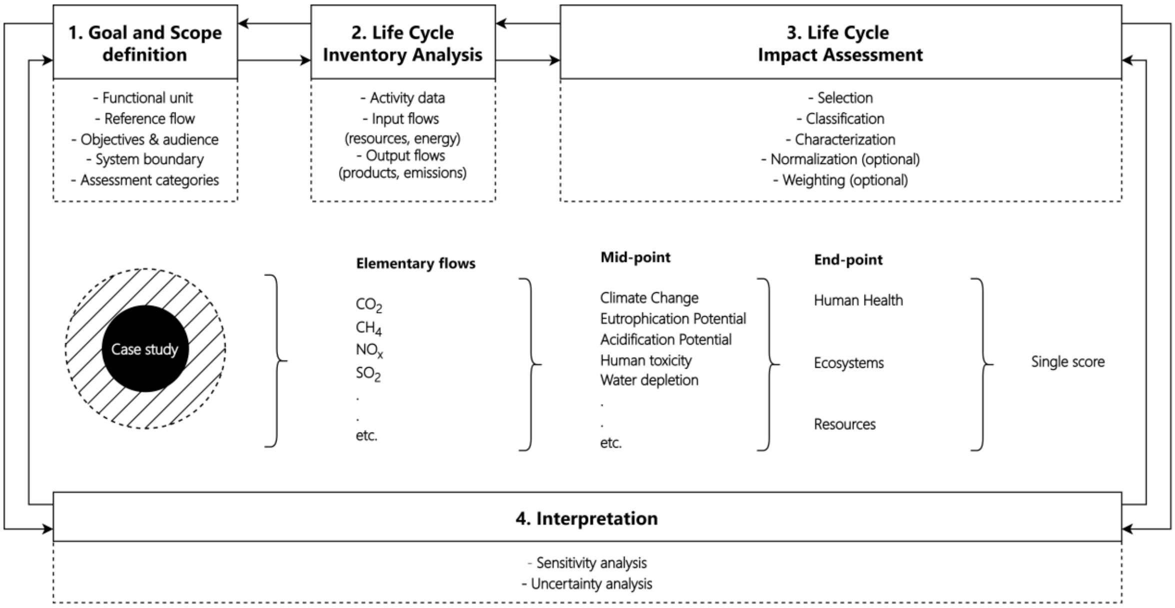


Figure 2-2. Life Cycle Assessment Framework according to the descriptions in ISO (2006b, 2006a).

The **first phase**, Goal and Scope definition, sets out the context, audience, and intentions of the study (ISO, 2006a). It defines the functional unit (FU), reference flow, system boundaries, impact categories, and impact assessment methods selected for their relevance to the product system and study goals (Hauschild *et al.*, 2017). It specifies cut-off criteria and approaches for handling multifunctionality (Hauschild *et al.*, 2017). The environmental impacts are reported based on the FU, which is selected according to the goal of the study and should reflect the quantifiable function of the analyzed system during a given timeframe (e.g., the yearly production of 1 kg of apples, 1 kg of protein, or 1 ha of land) (Matthews *et al.*, 2014). The system boundary determines the scope of the LCA. In other words, the processes included in the assessment for the selected FU can be from cradle-to-grave (e.g., from raw material extraction until end-of-life of the consumed food product) or cradle-to-gate (e.g., from raw material extraction until farm gate) (Hauschild *et al.*, 2018). Cut-off criteria specify the criteria by which certain processes may be excluded from the assessment (e.g., if certain flows contribute less than 1% of the total impacts) (Matthews *et al.*, 2014). Multifunctional processes are systems yielding multiple products. To deal with multifunctionality issues, ISO (2006b) recommends a hierarchy of solutions: (i) subdividing production systems, (ii) system expansion to account for secondary functions, and (iii) allocation based on physical relationships (e.g., mass or economic attributes).

The **second phase**, Life Cycle Inventory Analysis (LCI) describes emissions, material, and energy flows (Hauschild *et al.*, 2017). This phase involves data collection for the product system, accounting for inputs and outputs (e.g., energy and material flows) and emissions to the air, water, and soil compartment associated with each phase of the product's life cycle (e.g., from raw material extraction to disposal) (ISO, 2006a). Foreground processes use specific data (e.g., primary data), while background processes rely on LCA databases (e.g., generic data, industry averages) (Jolliet *et al.*, 2017).

The **third phase**, Life Cycle Impact Assessment (LCIA), selects, classifies, and characterizes the inventory data into potential environmental and human health impacts (Hauschild *et al.*, 2017). The elementary flows are selected and then classified into a particular impact category (e.g., CH_4 is assigned to the Global Warming Potential impact category) (Jolliet *et al.*, 2017). The relative importance of each elementary flow within an impact category, to which it has been classified, is then 'characterized' (Jolliet *et al.*, 2017). This refers to their conversion into a common unit representative of the selected impact category using a characterization factor (e.g., CH_4 is converted to CO_2-equivalents by using a characterization factor of 29.8, since the Global Warming Potential of CH_4 is 29.8 times greater than CO_2) (Rosenbaum *et al.*, 2017; IPCC, 2021). Characterization factors are derived from environmental cause-effect models that estimate the potential impact of each emission, considering how substances move and affect ecosystems or human health after release (Rosenbaum *et al.*, 2017). They capture the cumulative impact and gradual decline of a specific emission over time (instead of only the immediate effects such as a sudden spike in pollution during a specific year) (Heijungs, 1995; Ryberg *et al.*, 2018).

Environmental impacts in this phase can be assessed at the midpoint or endpoint level. These levels represent different stages in the cause-effect chain that connect elementary flows to their environmental damage or impact category (Rosenbaum, 2017). For example, the midpoint level focuses on environmental impacts at an earlier stage and quantifies them in terms of specific categories (e.g., Global Warming Potential or Eutrophication Potential) (Jolliet *et al.*, 2017). It involves fewer assumptions and modeling steps, reducing uncertainty. The endpoint level goes further along the cause-effect chain and links environmental impacts to three areas of protection (i.e., Human Health, Ecosystems, and Resources) (Jolliet *et al.*, 2017). It translates and aggregates mid-point impact categories into final damages, increasing uncertainty. An optional step in the LCIA phase is to apply normalization and weighting (Stranddorf *et al.*, 2005; Laurent and Hauschild, 2015). During normalization, the characterized results can be adjusted against a reference system (e.g., average yearly emissions per person), converting all impact categories into a common metric (Laurent and Hauschild, 2015). To compare normalized values, a weighting step adjusts each category's severity into a single score based on value judgments (e.g., policy considerations) (Stranddorf *et al.*, 2005; Itsubo, 2015).

The **fourth phase**, Interpretation, interprets the results from the previous phases to make informed conclusions and recommendations (Hauschild *et al.*, 2017). This phase critically evaluates the LCI and LCIA phases and tests the strength and robustness of its conclusions by applying sensitivity or uncertainty analysis. Results obtained from the LCA are interpreted as 'potential' impacts because they are relative to the chosen functional unit and represent broad estimates that rely on aggregated impacts across various locations and time frames (assuming constant conditions, i.e., a steady state) (Rosenbaum *et al.*, 2018; Ryberg *et al.*, 2018).

2.3. Research design

Following the pragmatic research paradigm, this PhD used a mixed-method approach (Creswell, 2009; Hennink *et al.*, 2020), including an iterative cycle of inductive and deductive thinking (i.e., moves back and forth the design, sampling, and analytical cycles) (**Figure 2-3**). For example, when estimations deviated from established values in the literature (e.g., typical manure excretion rates for hens), observations specific to the farm level (e.g., variations in animal breed or management activities such as feed sources) were used to theorize potential causes for the deviations. The LCA model was then refined based on these insights, and the updated results were tested to see if they aligned more closely with theoretical expectations regarding the environmental impact of AFS (e.g., lower carbon footprint relative to a reference system). This approach was combined with an exploratory case study design (Yin, 2009; Phelan, 2011) to identify patterns and examine the complex interactions of AFS.

The overall research design followed three cycles which were adapted and modified from Hennink *et al.* (2020). The design cycle focused on addressing the objectives of the PhD by formulating specific research questions based on knowledge gaps identified in the literature (**Chapter 1**). The sampling cycle involved collecting data through various methods (e.g., semi-structured interviews, literature, surveys, triangulation, and field observations) and developing protocols (e.g., inclusion/exclusion criteria and case study selection), all aligned with the overall methodological framework (**Chapter 2**). The analytical cycle encompassed data curation, validation, and modeling, following the methodological framework, in order to draw conclusions and interpretations (**Chapter 3, Chapter 4,** and **Chapter 5**).

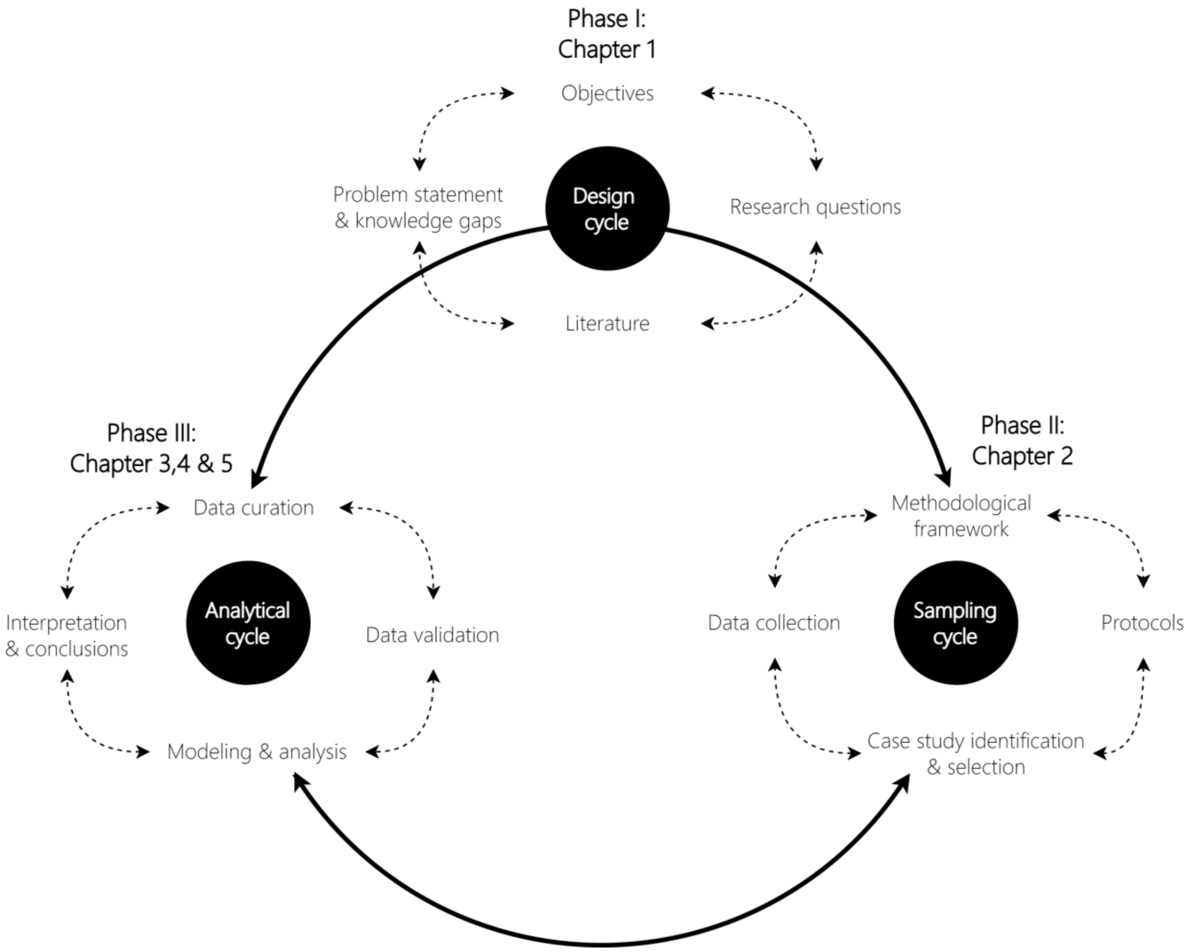


Figure 2-3. General research design and design cycles (black circles) for each chapter of the PhD thesis adapting partly the Hennink *et al.* (2020) research cycle.

Furthermore, the research design consisted of case studies representing AFS in a real-world context, each utilizing up to 32 individual studies and up to two farms as the unit of analysis and observation (UAO) (**Figure 2-4**). The selected UAOs were drawn from a pool of studies identified systematically in the literature (**Paper I**) and from a predefined pool of farms established via farmer networks (**Paper II** and **Paper III**).

The UAO in Austria and Portugal were chosen in dialogue with the local network partners from a pool of farms based on the following criteria:

(i) **Available and accessible data:** Farms that were accessible and had comprehensive datasets were chosen to minimize inconsistencies from incomplete or poor-quality data.

(ii) **Presence of only agri-food economic activities:** For consistency and to align with the general theme of this thesis, farms engaged only in agri-food production were selected to avoid interference from non-agricultural activities (e.g., tourism) that could influence the interpretation of the findings.

(iii) **Establishment of a clear silvopastoral configuration:** Ensuring that each UAO followed a well-defined silvopastoral configuration was important for enabling cross-case comparisons given that AFS can vary widely in design and function.

(iv) **Use of a single animal species per agroforestry land unit:** Focusing on only one animal species allowed for a more controlled analysis of the environmental impacts. Different animal species have distinct feeding habits, manure excretion rates, and overall environmental effects, which could complicate the LCA if multiple species were involved.

For **Paper I** and **Paper II**, an embedded single-case study design (Yin, 2009) was used. In **Paper I**, multiple individual studies were used as UAOs to explore the environmental performance of AFS in food production within a global context, with several agroforestry configurations representing the overarching case. In **Paper II**, an in-depth analysis was conducted on the environmental performance of two contrasting farms as UAOs in an Austrian context, under the broader case of silvopasture. In **Paper III** a holistic single-case study approach (Yin, 2009) was used, which focused on a single UAO in a Portuguese context, also using silvopasture as the overarching case. A multiple-case study design (Yin, 2009) was applied across the three papers, (i.e., integrating their respective designs) to draw broader cross-case limitations, perspectives, and conclusions in **Chapter 4,** and **Chapter 5**. This approach allowed comparisons between the UAOs by extracting insights presented in each paper, enabling the identification of patterns, similarities, and differences. An illustrative summary is shown in **Figure 2-4**.

An attributional LCA (ALCA) approach was applied throughout this thesis, focusing specifically on the environmental domain. ALCA provides a snapshot of the system's environmental impacts without considering the broader market-driven changes that could result from shifts in production or consumption patterns (Sandin *et al.*, 2016). The main environmental indicators explored in this thesis (**Paper II**), along with their relevance to AFS, are outlined below:

(i) **Climate Change or Carbon Footprint (CF) in CO_2-eq:** CF refers to the global warming potential, measuring the net impact of GHG (such as CO_2 and CH_4) on climate change over time (e.g., 100-year period). AFS can both emit GHG and sequester carbon, thus influencing net emissions. Research highlights that some AFS can potentially mitigate climate change by capturing carbon in the woody biomass and soils, though the extent varies by system and region (Ramachandran Nair and Toth, 2016; Quevedo-Cascante *et al.*, 2023).

(ii) **Eutrophication Potential (EP) in PO_4-eq:** EP measures the risk of water bodies being over-enriched with nutrients like nitrogen and phosphorus, leading to algal blooms and oxygen depletion. AFS may reduce nutrient runoff through nutrient retention, but they can also contribute to nutrient leaching depending on management practices (Jose, 2009; Schroth *et al.*, 2016; Manevski *et al.*, 2019).

(iii) **Acidification Potential (AP) in SO_2-eq:** AP quantifies the release of acidifying substances, such as sulfur dioxide (SO_2) and nitrogen oxides (NO_X), which can harm ecosystems through acid rain. In AFS, the use of mineral fertilizers and manure can contribute to acidifying emissions. Nevertheless, it has been documented that the presence of trees can help mitigate emissions (Sollen-Norrlin *et al.*, 2020).

(iv) **Land Occupation (LO) in m^2:** LO refers to the amount of land used over a specific period. AFS often provide multiple outputs, such as timber and crops. This makes land-use efficiency an important factor when evaluating AFS (García de Jalón *et al.*, 2018).

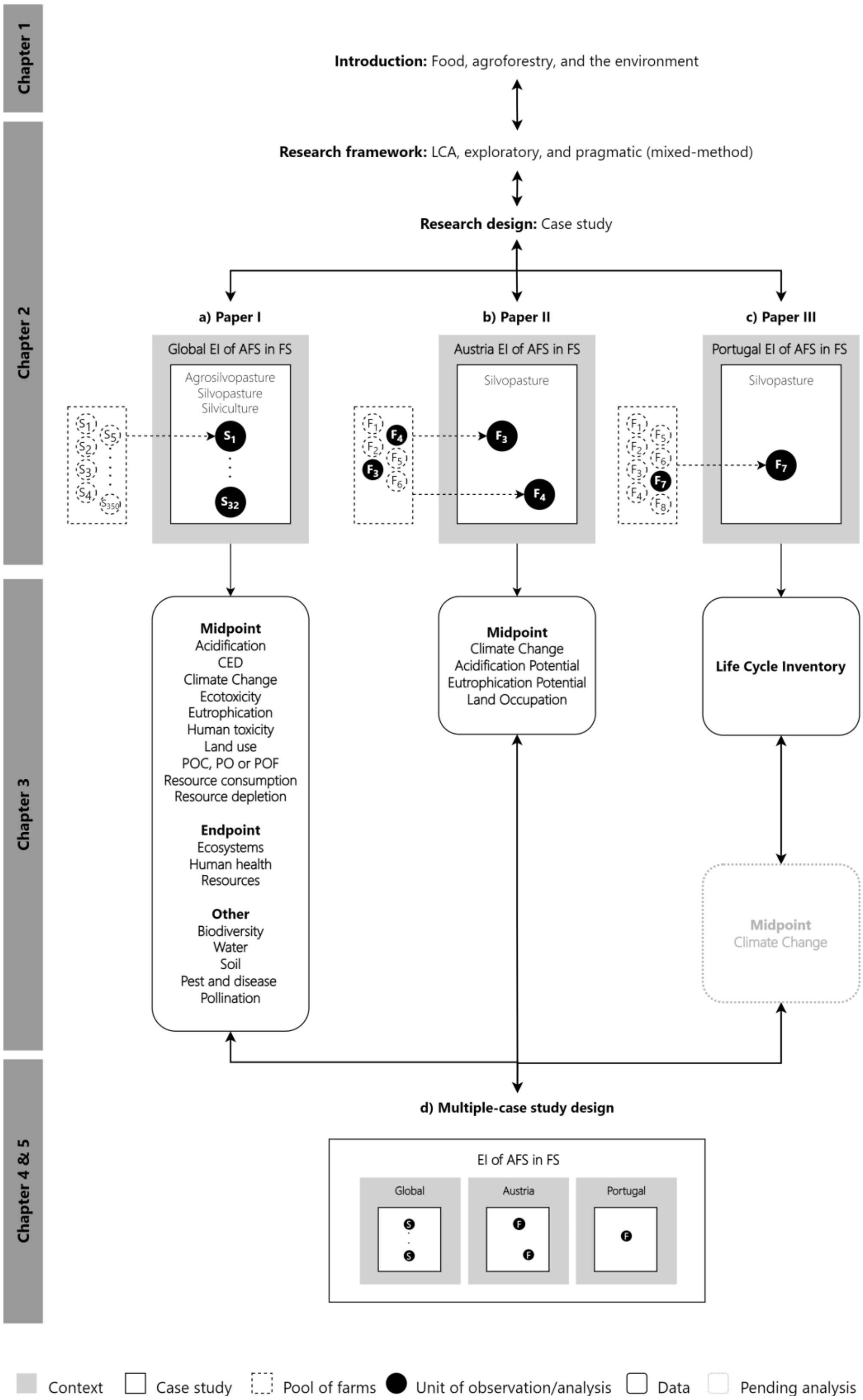
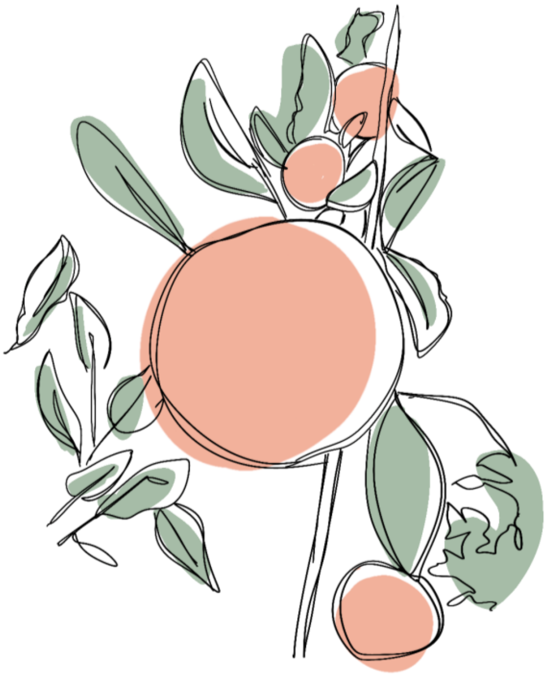


Figure 2-4. Case study design and thesis structure where a) embedded and single-case study design (**Paper I**), b) embedded and single-case study design (**Paper II**), c) holistic and single-case study design (**Paper III**), and d) the respective design of each paper is then integrated as a multiple-case study design to draw broader conclusions regarding the environmental impacts (EI) of agroforestry systems (AFS) in food systems (FS). The dashed lines and dashed circles represent the pool of studies (S) and farms (F) from which the units of observation/analysis (in black) were selected.



Chapter 3

3. Results

This chapter is divided into three sections corresponding to two complete scientific papers and one preliminary scientific draft. It provides summarized information regarding the objectives, methodological approach, and main results of the three papers. The complete outline and supplementary materials for **Paper I**, **Paper II**, and **Paper III** are presented in **Appendix A, Appendix B,** and **Appendix C**, respectively.

3.1. Paper I [Published]

3.1.1. Objective

Paper I systematically reviews how Life Cycle Assessment (LCA) has been applied to agroforestry in the context of food systems. The study aimed to (i) identify existing LCA studies on agroforestry, (ii) analyze the methodological choices made across the four phases of LCA and their respective agroforestry configuration, and (iii) evaluate the extent to which LCAs capture key environmental outcomes related to the general agroforestry literature and their respective agroforestry configuration.

3.1.2. Methodology

A systematic review approach of peer-reviewed LCA studies related to agroforestry systems (AFS) was used (**Figure 3-1**). The process involved a comprehensive literature search using databases such as Web of Science, Scopus, AGRIS, and CAB Direct. The search terms focused on LCA methodologies and definitions of AFS. Studies were included in the review based on relevance to food production, their focus on agroforestry practices, and explicit use of the LCA methodology. Only primary research articles written in English were included. From a pool of 350 studies, 32 relevant LCA studies were identified. Data were extracted and categorized into multiple themes. The paper synthesized both qualitative and quantitative data, analyzing each phase of the LCA framework and various environmental outcomes defined in the general agroforestry literature across the selected LCAs.

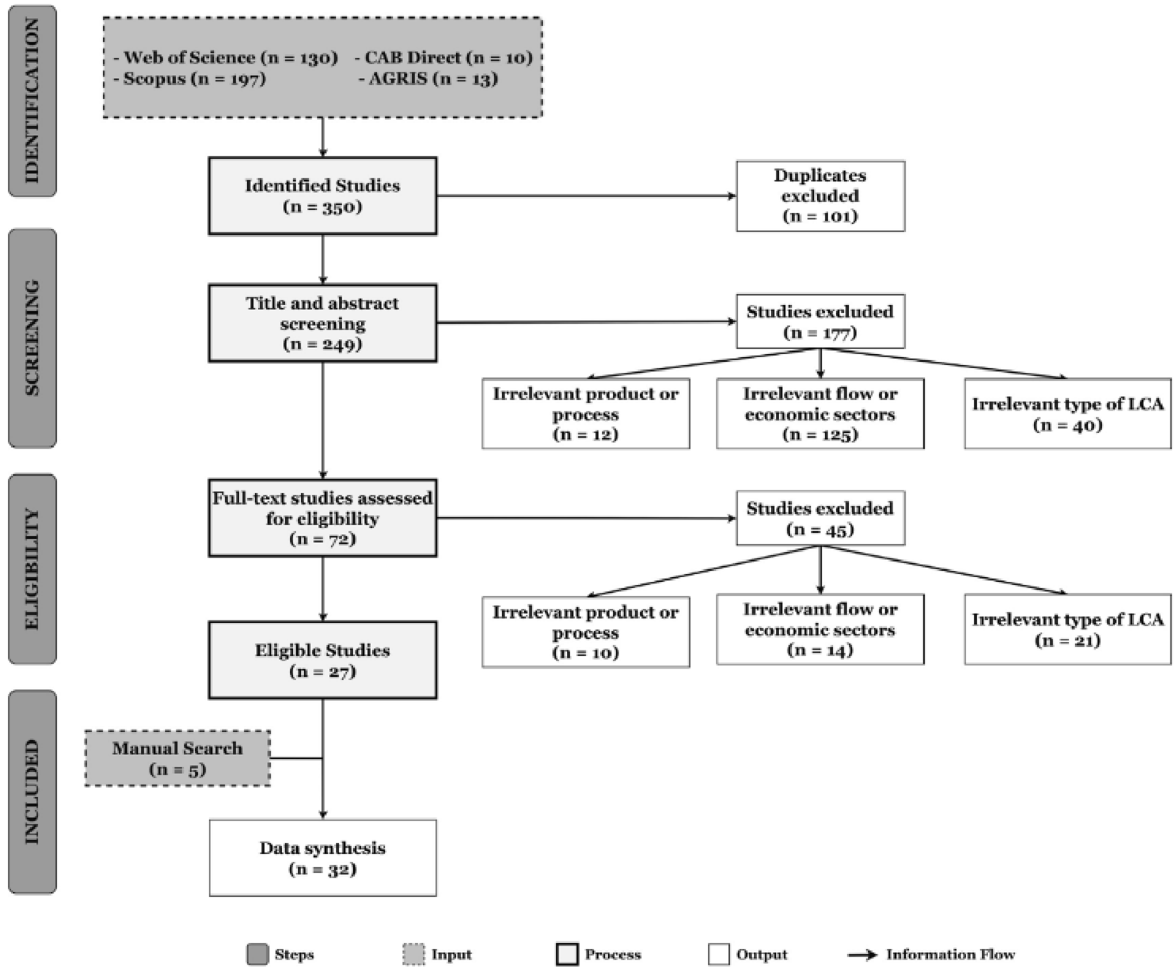


Figure 3-1. Systematic review process (Quevedo-Cascante *et al.*, 2023).

3.1.3. Main results

The results of the systematic review were organized according to the four phases of the LCA framework. In the **goal and scope definition phase**, the review showed a common trend in how functional units are applied across the studies. The majority use a mass-based functional unit, particularly in agrosilvicultural systems, with fewer studies adopting area-based or economic-based units. The system boundaries were typically set at cradle-to-farm gate, with limited consideration of post-farm gate stages like processing, retail, and consumption. In addition, nearly half of the reviewed LCA studies were focused on tropical regions, while the rest were concentrated on temperate climates, predominantly in Southern Europe. The agroforestry practices assessed were mainly agrosilvicultural and then agrosilvopastoral and silvopastoral.

In the **life cycle inventory phase**, the review identified that most studies did not explicitly address multifunctionality. Where multifunctionality was considered, the allocation of impacts was typically based on physical or economic properties, with limited use of other allocation methods (e.g., biophysical allocation).

For the **life cycle impact assessment phase** (**Table 3-1**), a total of 18 studies focused only on climate change, while 14 assessed multiple impact categories. Overall, 30 studies conducted their assessments at the midpoint level, with 4 using endpoint-level analysis. At the midpoint, climate change was the most frequently assessed, appearing in all 30 studies. Eutrophication was included in 37% of studies, and acidification in 33%. Other impact categories like ecotoxicity and resource depletion were covered in 7 studies each, while Photochemical Oxidant Creation (POC) or Photochemical Oxidation (PO) were addressed in 6 studies. Less frequently assessed impacts were Cumulative Energy Demand (CED) and human toxicity (both in 4 studies), and land use (3 studies). At the endpoint level, human health, ecosystems, and resources were equally considered across 4 studies. Only two studies conducted their assessment both at the midpoint and endpoint level.

Table 3-1. Frequency of midpoint and endpoint impact indicators evaluated across selected studies, per agroforestry configuration and overall. A dark color code means that the number of studies considering this impact category is above the average of the total number of studies for all agroforestry configurations (Quevedo-Cascante *et al.*, 2023).

Categories[a]	Agrosilvicultural (*n* = 13)	Agrosilvopastoral (n = 11)	Silvopastoral (n = 8)	Total
Midpoint				
Acidification	6	4	1	10
CED	2	2	0	4
Climate Change	13	11	8	30
Ecotoxicity	4	4	0	7
Eutrophication	7	4	2	11
Human toxicity	2	2	0	4
Land use	1	2	0	3
POC, PO or POF	3	3	0	6
Resource consumption	1	0	0	1
Resource depletion	4	4	0	7
Endpoint				
	Agrosilvicultural (n = 1)	**Agrosilvopastoral (n = 1)**	**Silvopastoral (n = 2)**	**Total**
Ecosystems	1	1	2	4
Human health	1	1	2	4
Resources	0	1	2	3

[a] Four 'Other LCIA' indicators (described in supplementary Table S4) used in four studies (Armengot et al., 2021; Costa et al., 2018; Livingstone et al., 2021; Rocchi et al., 2019) were not included in this table since neither a midpoint nor an endpoint method was applied.

Furthermore, there is substantial variability in the reported carbon footprint (CF) across and within 18 studies (**Figure 3-2**), which were categorized into three distinct production systems: milk, crop, and meat, totaling 82 observations. These were further grouped as silvopastoral (36 values), agrosilvopastoral (38 values), and agrosilvicultural (8 values). Meat production showed the highest variability, particularly with negative CF values in silvopastoral and agrosilvopastoral systems, mainly for beef and calf. For meat production systems, the CF was compiled for beef (−28 to 18 kg CO_2-eq/kg LW), calf (−22 to 31 kg CO_2-eq/kg LW), sheep (including lamb) (1 to 26 kg CO_2-eq/kg LW), and pig (−3 to 6 kg CO_2-eq/kg LW). For milk production systems, the CF was compiled for cow milk (0.5 to 3 kg CO_2-eq/kg FPCM) and goat milk (0 to 3 kg CO_2-eq/kg). In crop production, cocoa, coffee, and olive had CF values ranging from −0.03 to 4, 3 to 5, and 0.1 to 0.6 kg CO_2-eq/kg product, respectively.

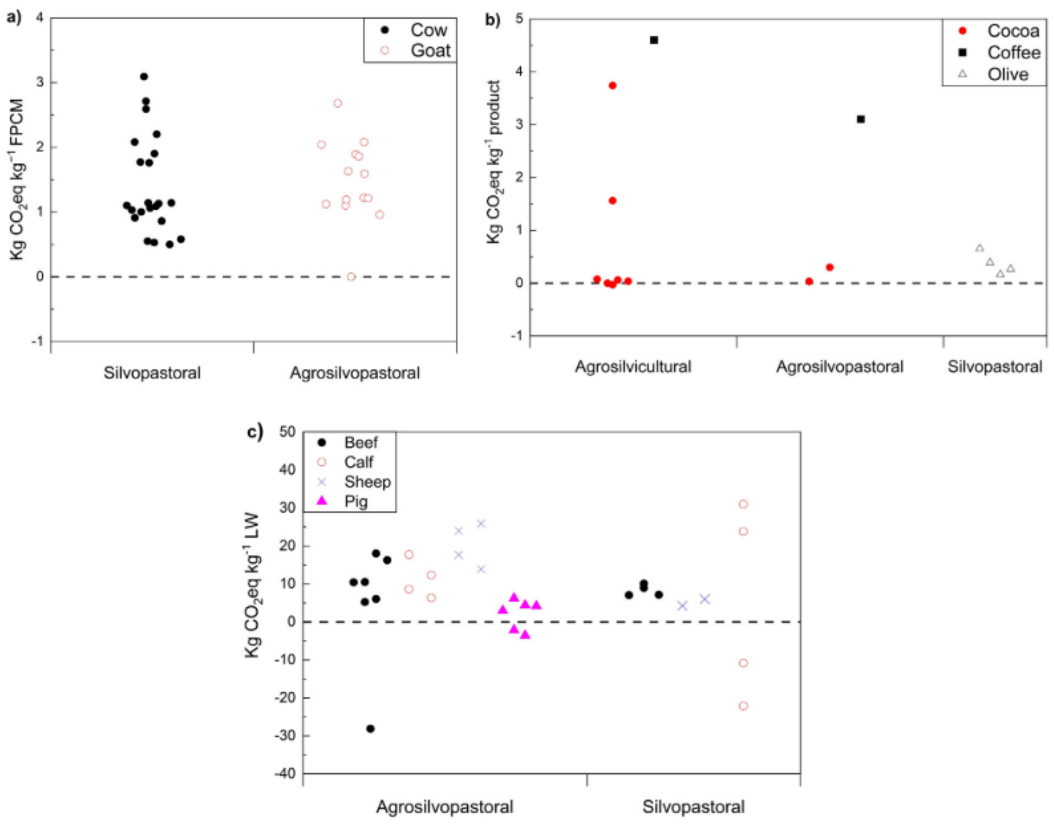


Figure 3-2. Scatter plots of climate change impact at midpoint level across three agroforestry configurations for three production systems (cradle-to-farm gate, mass units). a) Milk systems (kg CO_2-eq/kg FPCM): cow (black), goat (red). b) Crop systems (kg CO_2-eq/kg product): cocoa (red), coffee (black), olive (white). c) Meat systems (kg CO_2-eq/kg LW): beef (black), calf (red), sheep (blue), pig (pink) (Quevedo-Cascante *et al.*, 2023).

When examining six of the often-cited environmental outcomes of AFS in the broader agroforestry literature (i.e., non-LCA studies), most outcomes were not captured in LCA models or were underrepresented or absent in current studies (**Table 3-2**). The focus on climate change impacts, although important, presented an incomplete view of agroforestry's environmental performance, as other potential impacts, both positive and negative, were not included.

Table 3-2. Frequency of environmental outcomes in the general agroforestry literature analyzed in the selected studies (n = 32), per AFS and in total. A dark color code means that the number of studies considering this outcome is above the average of the total number of studies for all agroforestry configurations (Quevedo-Cascante *et al.*, 2023).

Environmental outcome	Agrosilvicultural	Agrosilvopastoral	Silvopastoral	Total
Biodiversity	1	2	2	5
Climate change mitigation	4	6	5	15
Water	2	4	0	6
Soil	1	1	0	2
Pest and disease control	0	0	0	0
Pollination	0	0	0	0

In the **interpretation phase**, sensitivity and uncertainty analyses were conducted in only a small portion of the studies, and those that did perform these analyses often focused on limited aspects, such as allocation methods or emission factors.

3.2. Paper II [Published]

3.2.1. Objective

This paper's objectives are twofold. First, it evaluates the environmental impacts and carbon footprint (CF) of two Austrian silvopastoral systems combining egg production within apple orchards. This analysis encompasses contribution analysis, including carbon sequestration (C-seq) potential, and post-harvest activities. Second, it examines two modeling approaches to manage multifunctionality at the farm gate. The models aim to assess methodological differences without representing a comprehensive set of LCA models or indicating preferences for any particular model. The LCA includes a comparison of the impacts of the agroforestry products against standard and specialized practices for apples (RS-A) and eggs (RS-E).

3.2.2. Methodology

A general overview of the methodological approach is shown in **Table 3-3**. A product-based FU was used for the reference year of 2021, namely "1 kg of fresh Class I apples" and "1 kg of fresh Class I eggs" from cradle-to-farm gate and cradle-to-retail both for each product in Farm 1 (F1) and Farm 2 (F2). Two modeling approaches were used for distributing the impacts of the entire farm system (until the farm gate) between agroforestry eggs and apples, where model 1 (M1) used an economic relationship and model 2 (M2) subdivision. The agroforestry food products were then compared to their reference system (RS-A for reference apples and RS-E for reference eggs).

Table 3-3. Summary of the methodological approach applied to paper II. LCI = Life Cycle Inventory; LCIA= Life Cycle Impact Assessment.

LCA phase		
1. Goal and scope definition	Objective	To assess the environmental and climate impacts of apples and eggs in two agroforestry farm systems (F1 and F2), to compare them against their reference system (RS-E and RS-A), and to test two modeling approaches at the farm gate.
	Functional unit	'1 kg fresh Class I apples' and '1 kg fresh Class I eggs'.
	System boundaries	Cradle-to-farm gate and cradle-to-retail.
	Impact categories	Climate Change (Carbon Footprint; CF) including C-seq and soil carbon (C) changes, Eutrophication Potential (EP), Acidification Potential (AP), Land occupation (LO).
2. LCI	Data	Flock productivity (population, breed, age, housing, egg yield, excretion, mortality, losses) Feed intake (concentrate and forage) Apple productivity (yield, fertilizers, density, between row management, biomass) Field operations (fertilizing, irrigation, maintenance, plant protection, harvesting, uprooting) Sorting, storage, and packaging operations Retail operations
	Multifunctionality	Model 1: Economic relationship Model 2: Sub-division Co-products: Economic allocation
	Estimation of emissions	N_2O, NH_3, NO_3, NO_X/NO_2
3. LCIA	Level	Mid-point
	Method	CML
4. Interpretation	Sensitivity analysis	Manure classification Land occupation C-sequestration

3.2.3. Main results

The cumulative impact per farm, consistent across both models, is illustrated in **Figure 3-3**.

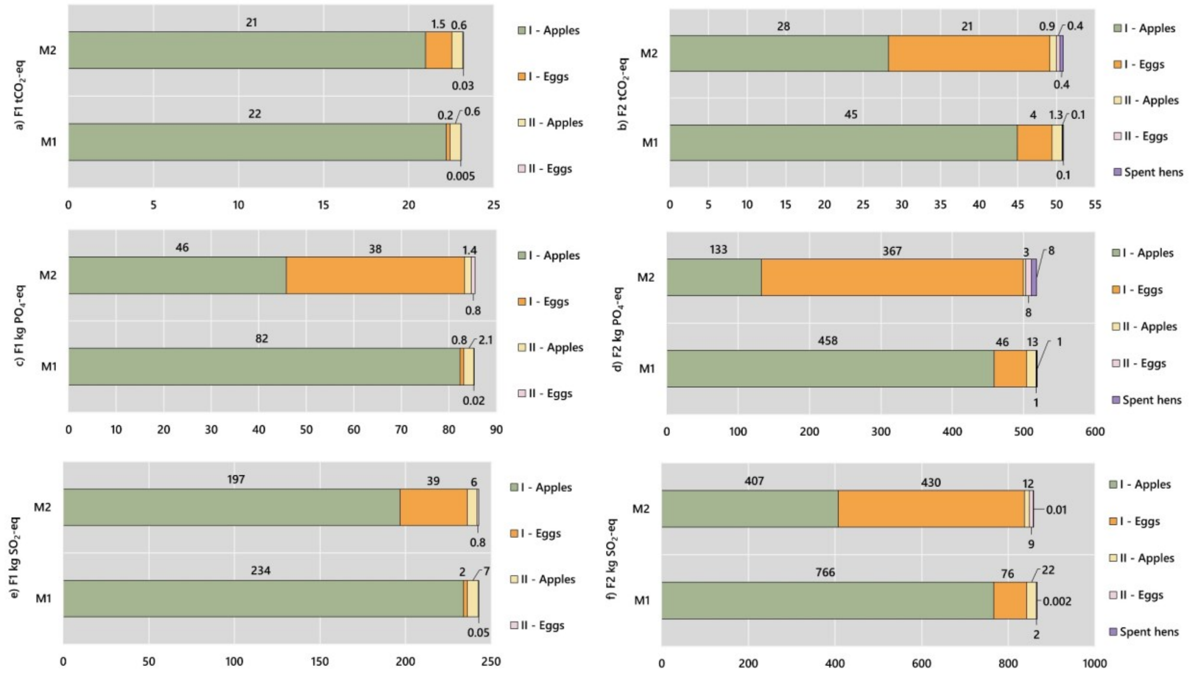


Figure 3-3. The distribution of total farm emissions across Class I products (I - Apples; I - Eggs), Class II products (II - Apples; II - Eggs), and additional co-products (spent hens) for Farm 1 (F1) and Farm 2 (F2). Emission metrics include a) and b) carbon footprint (t CO_2-eq), c) and d) eutrophication potential (kg PO_4-eq), and e) and f) acidification potential (kg SO_2-eq). Small variations between models result from rounding during data handling (Quevedo-Cascante, 2024).

Overall, the environmental impacts per kg of egg or apple from farms F1 and F2 are generally lower across most impact categories when compared to their reference systems. At the farm gate, CF for apples ranged from 0.09–0.17 kg CO_2-eq/kg, while for eggs it spanned 0.19–1.62 kg CO_2-eq/kg across all systems and models (**Figure 3-4**). C-seq reduced emissions by 22–42% for apples and by 0.4–39% for eggs, primarily due to the carbon contributions from plant biomass in apple production (84–99%), with manure contributing between 0.7–9%. EP varied between 0.19–1.7 g PO_4-eq/kg for apples and 0.7–35 g PO_4-eq/kg for eggs (**Figure 3-5**), while AP ranged from 0.8–2.9 g SO_2-eq/kg for apples and 2–36 g SO_2-eq/kg for eggs across all systems and models (**Figure 3-6**). LO spanned 0.3–0.6 m²/kg for apples and 0.8–9 m²/kg for eggs throughout the analyzed systems and models (**Figure 3-7**).

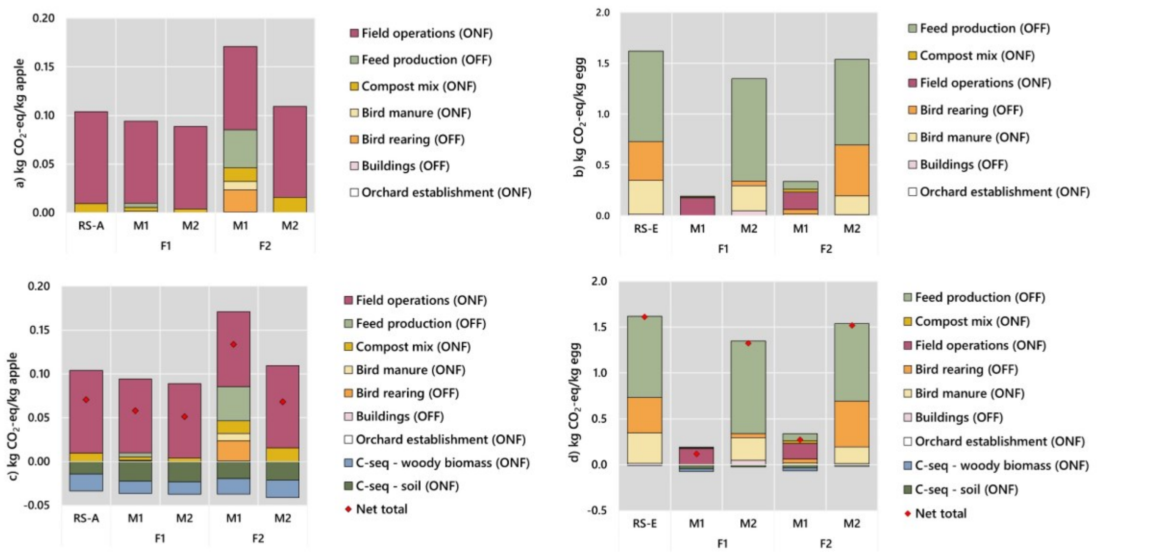


Figure 3-4. Carbon footprint per kg of apple (left) and egg (right) in kg CO_2-eq using two models (M1 and M2) for Farm 1 (F1), Farm 2 (F2), and reference systems for apples (RS-A) and eggs (RS-E), with results displayed both without C-seq contributions (top) and as net CF, incorporating soil C changes and C-seq in woody biomass (below) (Quevedo-Cascante, 2024).

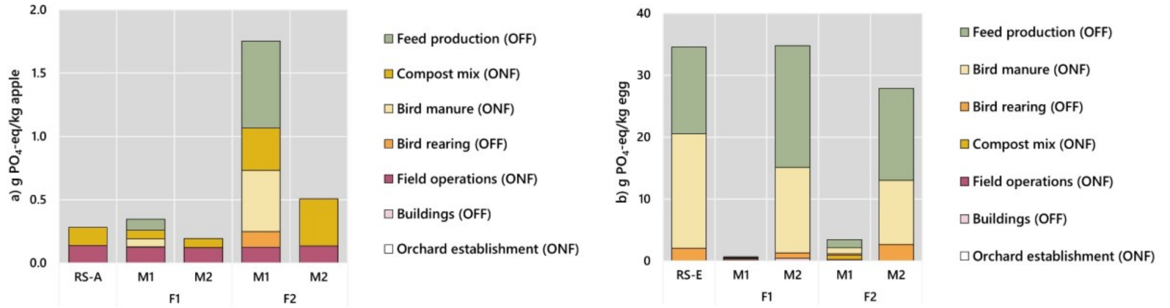


Figure 3-5. Eutrophication potential per kg of apple (left) and egg (right) in kg PO_4-eq, also using M1 and M2 for Farm 1 (F1), Farm 2 (F2), and reference systems for apples (RS-A) and eggs (RS-E) (Quevedo-Cascante, 2024).

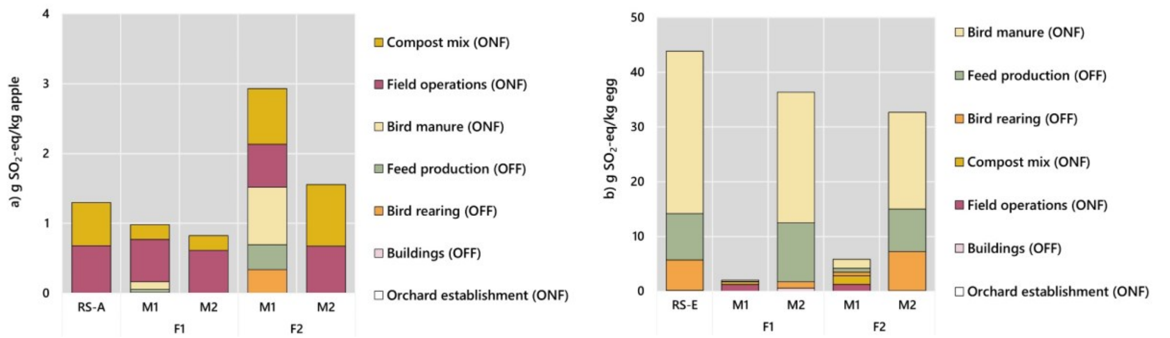


Figure 3-6. Acidification potential per kg of apple (left) and egg (right) was expressed in kg SO_2-eq, for the two modeling approaches (M1, M2) for Farm 1 (F1), Farm 2 (F2), and reference systems for apples (RS-A) and eggs (RS-E) (Quevedo-Cascante, 2024).

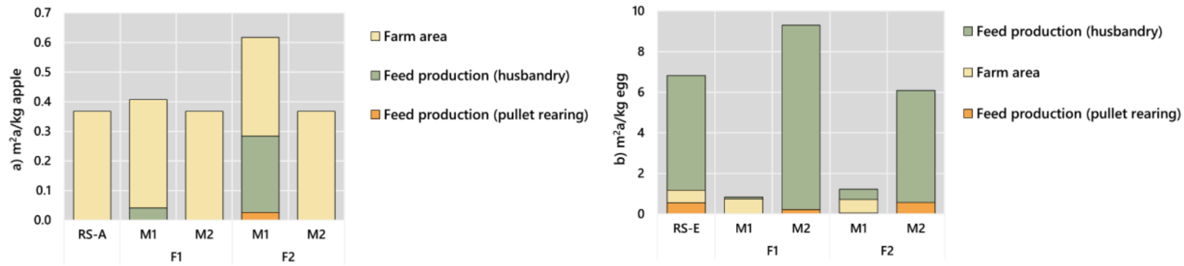


Figure 3-7. Land occupation per kg of apple (left) and egg (right) was measured in m^{2}a across two modeling approaches (M1, M2) for Farm 1 (F1), Farm 2 (F2), and reference systems for apples (RS-A) and eggs (RS-E) (Quevedo-Cascante, 2024)

Results show notable variation across models. M1 had consistently higher impacts for apples and significantly lower impacts for eggs in comparison to M2. In M1, apples carry a larger share of emissions due to their higher economic value, leading to a reduced CF, EP, AP, and LO per kg of eggs. In contrast, M2 uses subdivision, so emissions are driven by management practices, with differences between F1 and F2 primarily due to fertilization rates, feed production, and manure excretion rates. F1 benefits from reduced fertilization and use of spent hens, while F2 shows higher impacts from pullets, outdoor manure excretion, and greater feed consumption.

Post-harvest activities contributed up to 29% of the total EP and AP impacts and as much as 57% of the CF from cradle-to-retail. Overall, impacts per kg of egg or apple in F1 and F2 were generally lower across most categories compared to their reference systems, largely influenced by management practices and the production phase within the value chain.

3.3. Paper III [Draft]

3.3.1. Objective

This paper aims to establish a foreground Life Cycle Inventory (LCI) for a representative silvopastoral system in Portugal's naturally regenerated *Montado*, focusing on beef cattle and cork production from cradle-to-retail.

3.3.2. Methodology

Data collection combined local partner collaboration and literature for Montado-based silvopastoral systems in Portugal. Using a MIXED project survey, farm data was pre-collected and validated with interviews and observations, focusing on beef and cork in the Alentejo region. For cattle, LCI data covered both agroforestry and non-agroforestry farm activities, such as cow-calf operations, fattening activities, feed inputs, and live weight to retail. Cork data included tree density, agricultural inputs, and processing stages from cradle-to-retail. Upstream inputs like fertilizers and transport logistics, along with post-farm gate activities, were supplemented from literature and verified with local sources.

3.3.3. Preliminary results

The Montado silvopastoral system is an extensive, conventionally managed farm focused on cork and beef from cow-calf operations (the primary income source). The system boundary from cradle-to-retail is shown in (**Figure 3-8**). For cork production (**Table 3-4**), stripping begins when trees are 30 years old and recur every nine years using axes with an estimated yield of 5 t fresh weight (FW)/yr or 9.8 kg/ha. Cork oak comprises 95% of trees, with 5% holm oak, and is part of a naturally regenerating and rainfed landscape. Stand establishment practices common in Portuguese systems, such as planting and fertilizing, are excluded, as is pruning or thinning. Diesel use relates mainly to worker transport and cork striping (transportation of the harvested material). Raw cork is stored on-site, processed locally, and sold as cork stoppers to international and national retail markets.

Table 3-4. General characteristics, and input and output data for cork production until farm-gate (Quevedo-Cascante, 2024, paper III draft manuscript)

Parameter	Unit	Value
General characteristics		
Plantation	Type	Natural
Productive tree species	Type	Cork oak
Other tree species	Type	Holm oak
Farmland	ha	681
Grassland (silvopasture)	ha	508
Arable (non-agroforestry)	ha	70
Other (other agroforestry)[a]	ha	103
Total tree density	#/ha	5
Cork oak	#/ha	4.75
Holm oak	#/ha	0.25
Tree height	m	5
Harvest cycle	Years	9
Harvest	#/yr	0.1
INPUT		

Machinery use (cork stripping)	hr/yr	22.2
Workers (traveling to the farm)	hr/yr	25
Fences (maintenance)	hr/yr	26.67
Seeds input	kg/ha	0
Fertilizer N, P, K	kg/ha	0
Limestone	kg/ha	0
Irrigation	m^3/ha	0
Herbicide applications	#/yr	0
Fungicides applications	#/yr	0
Insecticides applications	#/yr	0
Pruning	#/yr	0
OUTPUT		
Raw reproduction cork[b,c]	kg FM/yr	5000

[a]non-productive (buffer zones) and permanent crops
[b]2.3 €/kg (35 €/'arroba', where 1 'arroba'=15 kg)
[c]Total 3000 'arroba' (high-quality cork price)

The cattle system includes Limousine and Angus cows and heifers for replacement and slaughter, with a stocking density of 0.44 animals/ha (**Table 3-5**). Calves graze with cows until weaning, with manure left unmanaged on the soil. Feeding relies on grazing and browsing, with concentrates provided only to calves and occasionally to adults during feed scarcity. Calves, sold at 230 kg live weight (LW) to fattening farms, spend six months in the system, while cows reach 700 kg LW before slaughter. Most farms use regional fattening facilities where feed is concentrate-based. Fattened cattle are sold at 500 kg LW, and manure from fattening facilities supports arable land for roughage. Meat from local slaughterhouses is marketed domestically and internationally.

Table 3-5. General characteristics, and input and output data for beef production until farm-gate (Quevedo-Cascante, 2024, paper III draft manuscript)

Parameter	Unit	Value
General characteristics		
System	Type	Extensive
Management	Type	Conventional
Breed 1	Type	Limousine
Breed 2	Type	Angus
Calves	# born/cow/yr	0.92
Calving interval	Months	13
Manure management	Type	Left on the field
Stocking density	# animals/ha	0.44
Grazing	# days/yr	375
Grazing	# hr/day	24
Grazing	Type	Continuous

Grass	Type	Grass clover
Legume proportion (grass)	%	50-75
Mortality rate	%	2
INPUT		
Annual beef breed cow	#/yr	200
Annual beef breed bull for breeding	#/yr	3
Annual beef breed heifer for replacement	#/yr	29
Annual beef breed cow	kg LW/yr	600
Concentrate (calf)[a,b]	kg FW/day	4
Concentrate (bull)[b,c]	kg FW/day	7
Concentrates (cow)[b,c]	kg FW/day	4
Forage[d]	# bales/yr	600
OUTPUTs		
Average weight of sold calf[d]	kg LW/yr	230
Average weight of sold discarded cow[e]	kg LW/yr	700

[a]During 4 months period.
[b]Ingredients: Maize, Barley, Sunflower hulls, Wheat bran, Soya beans, Cane molasses, Lucerne, Hydrogenated Fat, Calcium Carbonate, Calcium Phosphate, Sodium Bicarbonate, Sepiolite, Trace elements.
[c]From August to November
[d]Silage produced in 33 ha of arable land (conventional)
[e]600 euros/calf (6 months of age)
[f]800 euros/discarded cow

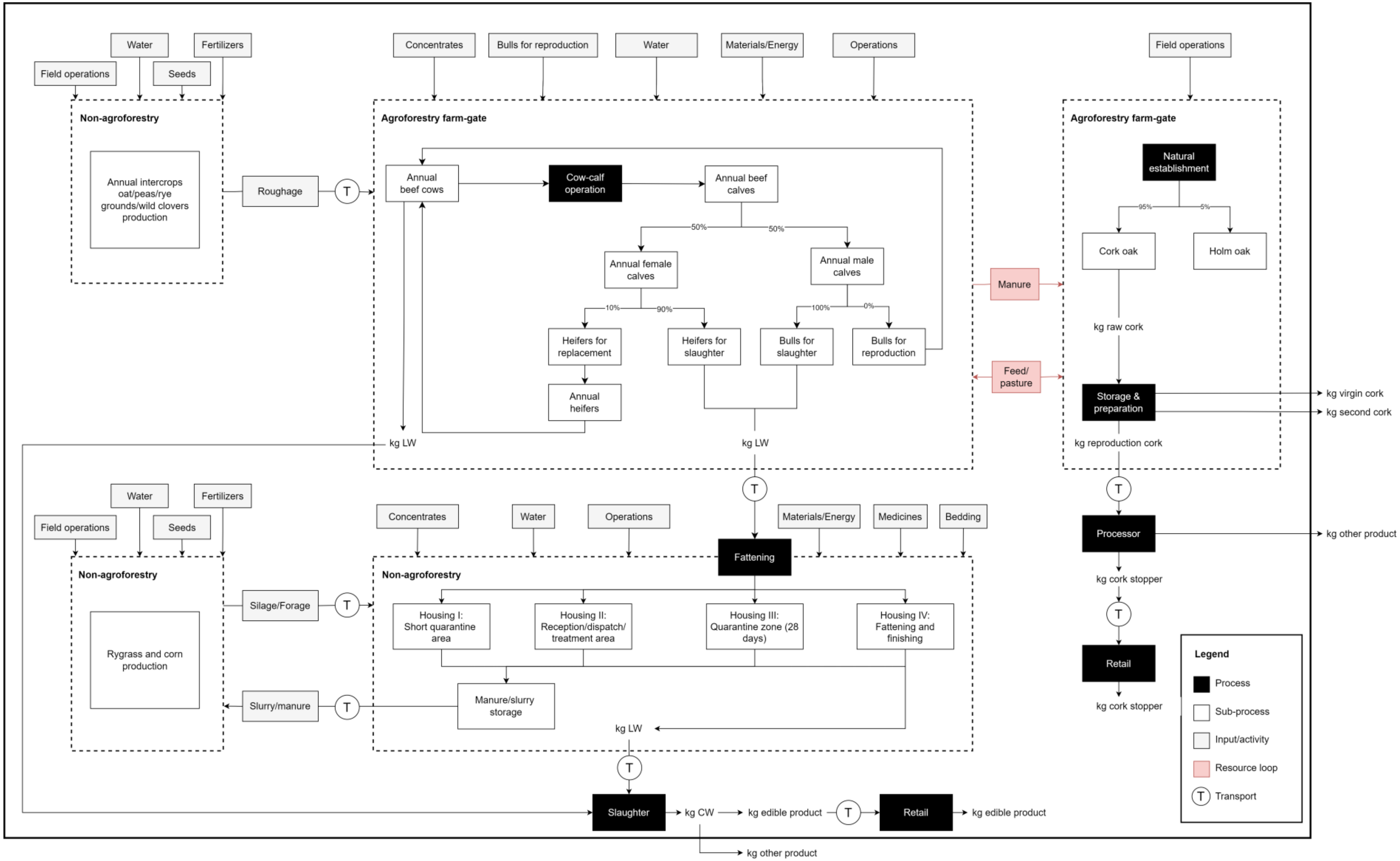


Figure 3-8. System boundary (in thick black square) of a representative silvopastoral system in the *Montado* from cradle-to-retail for cattle and cork production (Quevedo-Cascante, 2024, paper III draft manuscript).

Chapter 4

4. Discussion

This chapter provides a critical analysis of the research findings and the analyzed impact categories, including further research needs. The chapter is structured into four parts, with the first and second parts addressing the central research question of this thesis. The first part reflects on what insights can LCA provide when comparing the environmental and climate impacts of different food products from agroforestry, and the second part explores how different methodological choices influence the outcomes of LCAs. The third part addresses the constraints in the LCA framework and challenges with the chosen research design. Research perspectives and recommendations are then presented, identifying areas and research questions where further investigation is needed to enhance the understanding of agroforestry's role in the global food system.

4.1. Environmental and climate impacts

In addressing the first part of the central research question, *what insights can LCA provide when comparing the environmental and climate impacts of different food products from AFS?*, findings from **Paper I** and **Paper II** are further synthesized and compared against the non-agroforestry literature. Findings in **Paper I** indicate substantial variability in the reported CF across and within 18 agroforestry LCA studies. When comparing these AFS to average values from the non-AFS literature – derived from global systematic reviews, national averages, and case studies covering conventional, organic, and other farming approaches – significant differences are shown. The methodological choices leading partially to the reported results are discussed in **section 4.2**.

In crop production systems (CPS) (**Figure 4-1**), the CF for coffee in **Paper I** averages 3.9 kg CO_2-eq/kg, which is lower than the value reported for conventional systems (15 kg CO_2-eq/kg coffee), but slightly higher than the 'sustainable' systems (3.5 kg CO_2-eq/kg coffee) in Nab and Maslin (2020). A similar trend is observed for cocoa, where **Paper I** reports an average CF of 0.96 kg CO_2-eq/kg cocoa, significantly below the range of 1.67 to 6.76 kg CO_2-eq/kg cocoa observed in the systematic review by Wang and Dong (2024). For olive production, **Paper I** shows an average CF of 0.4 kg CO_2-eq/kg olive, which is slightly above the average 0.3 kg CO_2-eq/kg olive reported by Romero-Gámez *et al.* (2017) in their LCA of twelve Spanish olive systems, though within the study's reported range of 0.19 to 0.95 kg CO_2-eq/kg olive.

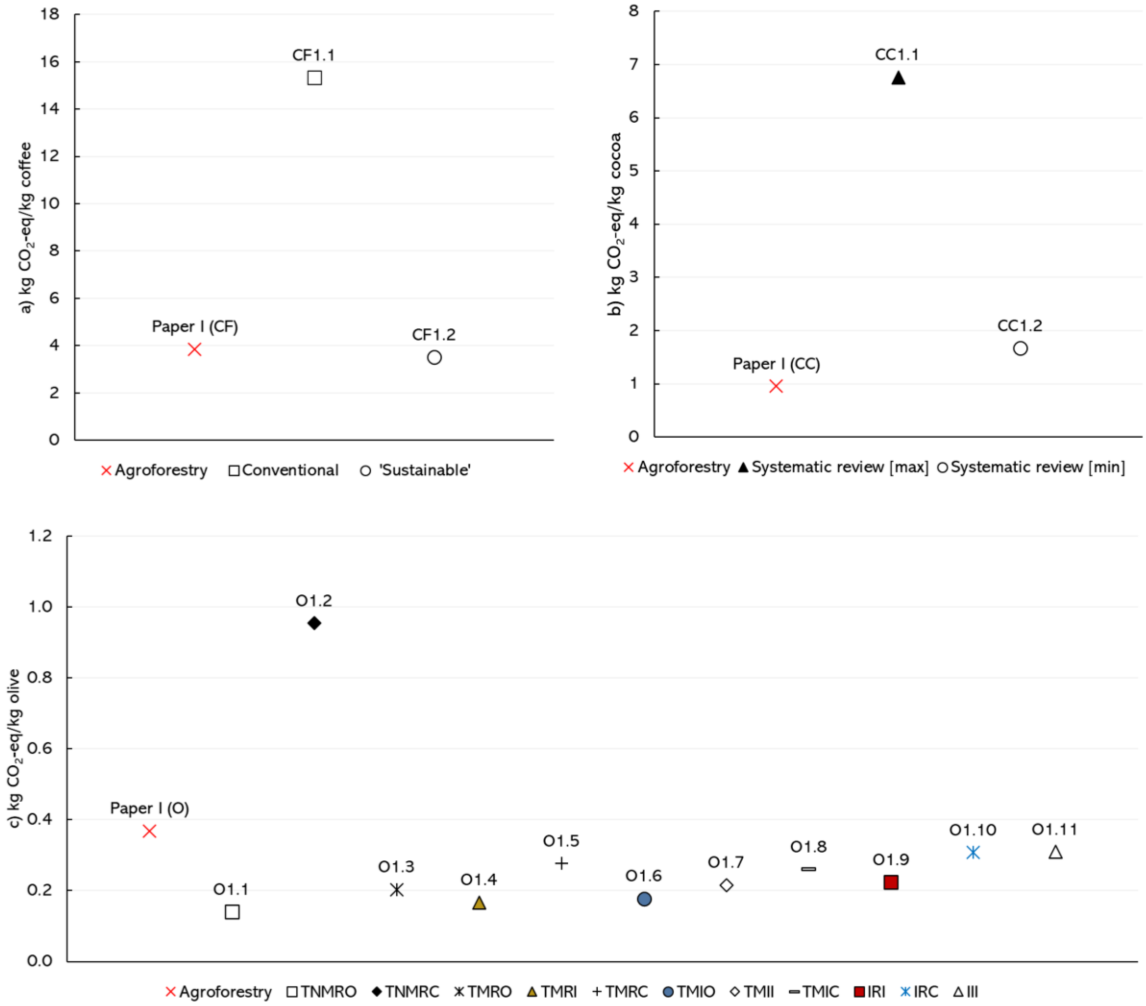


Figure 4-1. Scatter plots of the average Carbon Footprint (CF) for crop production system (CPS), as defined in **Paper I** from cradle-to-farm gate. Units in a) kg CO_2-eq/kg coffee, b) kg CO_2-eq/kg cocoa, c) kg CO_2-eq/kg olive. Data for coffee (CF): (Nab and Maslin, 2020). Data for cocoa (CC): (Wang and Dong, 2024). Data for olive (O): (Romero-Gámez *et al.*, 2017). Average data for agroforestry **Paper I**: (Quevedo-Cascante *et al.*, 2023). A number after the alphabet value indicates multiple observations in the same study. TNMRO: Traditional Non-Mechanized Rainfed Organic; TNMRC: Traditional Non-Mechanized Rainfed Conventional; TMRO: Traditional Mechanized Rainfed Organic; TMRC: Traditional Mechanized Rainfed Conventional; TMRI: Traditional Mechanized Rainfed Integrated; TMIO: Traditional Mechanized Irrigation Organic; TMII: Traditional Mechanized Irrigation Integrated; TMIC: Traditional Mechanized Irrigation Conventional; IRI: Intensive Mechanized Rainfed Integrated; IRC: Intensive Mechanized Rainfed Conventional; III: Super-Intensive Mechanized Irrigation Integrated; SII: Super-Intensive Mechanized Irrigation Integrated.

For milk production systems (MPS) from cradle to farm gate (**Figure 4-2**), the results between agroforestry and non-AFS are less divergent. In dairy goat production, **Paper I** findings show an average CF of 1.6 kg CO_2-eq/kg FPCM, which is higher than the average of 1.2 kg CO_2-eq/kg FPCM found in a Spanish comparative LCA conducted by Mancilla-Leytón *et al.* (2023). Conversely, in dairy cow production, the average CF in **Paper I** (1.6 kg CO_2-eq/kg FCPM) is lower than the global average of 2.1 kg CO_2-eq/kg FPCM documented in the systematic review by Mazzetto *et al.* (2022).

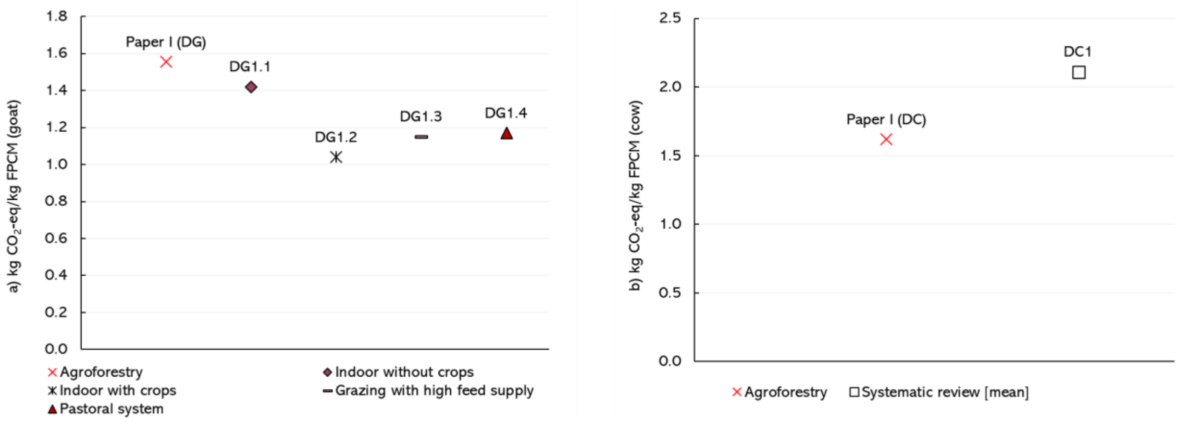


Figure 4-2. Scatter plots of the average Carbon Footprint (CF) for milk production system (MPS), as defined in **Paper I** from cradle to farm gate. Units in a) kg CO_2-eq/kg FPCM (goat) and b) kg CO_2-eq/kg FPCM (cow). Data for dairy goat (DG): (Mancilla-Leytón *et al.*, 2023). Data for dairy cow (DC): (Mazzetto *et al.*, 2022). Average data for agroforestry **Paper I**: (Quevedo-Cascante *et al.*, 2023). A number after the alphabet value indicates multiple observations in the same study.

For meat production systems (MTPS) (**Figure 4-3**), **Paper I** reports an average CF for beef of 16.5 kg CO_2-eq/kg LW, which is lower than the average of 18.1 kg CO_2-eq/kg LW reported for non-AFS, but with values ranging as high as 32.7 and 23.1 kg CO_2-eq/kg LW in global and country-specific reviews conducted by Hassan Pishgar-Komleh *et al.* (2022) and Mazzetto *et al.* (2021), respectively. In lamb production, however, **Paper I** on average has a value of 17.5 kg CO_2-eq/kg LW, which is higher than 14.6 kg CO_2-eq/kg LW noted in a summary report for sheep production in New Zealand and the United Kingdom (Mazzetto *et al.*, 2021). For pig production, **Paper I** shows an average CF of 4.1 kg CO_2-eq/kg LW, slightly above the mean 3.3 kg CO_2-eq/kg reported in a global systematic review of intensive pig systems by Zhang *et al.* (2024).

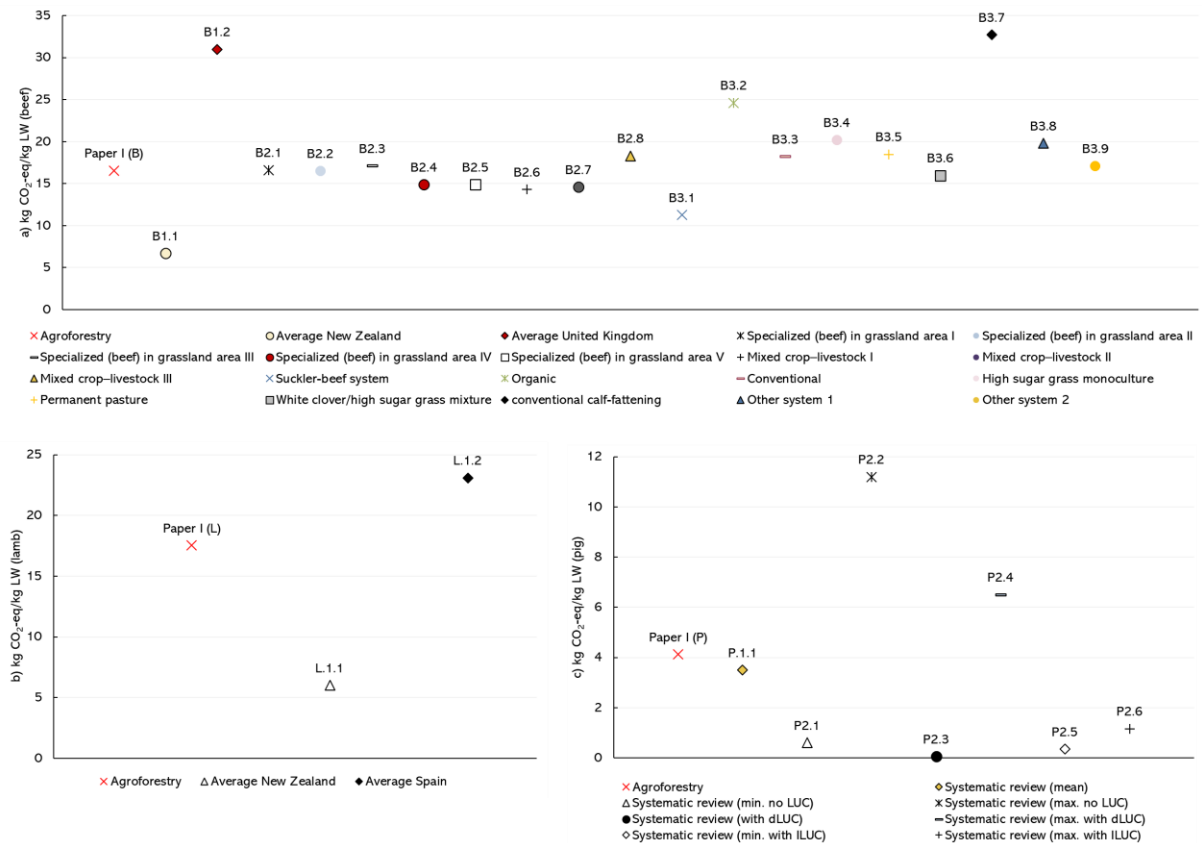


Figure 4-3. Scatter plots of the average Carbon Footprint (CF) for the meat production system (MTPS), as defined in **Paper I** from cradle to farm gate. Units in a) kg CO_2-eq/kg LW (beef), b) kg CO_2-eq/kg LW (lamb), c) kg CO_2-eq/kg LW (pig). Data for beef (B): (Veysset *et al.*, 2010; Mazzetto *et al.*, 2021; Pishgar-Komleh and Beldman, 2022). Data for lamb (L): (Mazzetto *et al.*, 2021). Data for pig (P): (Gislason *et al.*, 2023; Zhang *et al.*, 2024). Average data for agroforestry **Paper I**: (Quevedo-Cascante *et al.*, 2023). A number after the alphabet value indicates multiple observations in the same study. LUC: Land use change; iLUC: indirect land use change; dLUC: direct land use change.

Overall, the AFS from **Paper I** tends to show lower CF for certain crops and meat products, particularly for coffee, cocoa, and beef, when compared to non-AFS (**Figure 4-4**). Some reductions are observed in dairy systems, though dairy goat systems under agroforestry show slightly higher emissions. Similarly, while agroforestry practices demonstrate a reduction in emissions for beef production, the results for lamb and pig production are more variable, with average emissions higher than those reported for non-AFS.

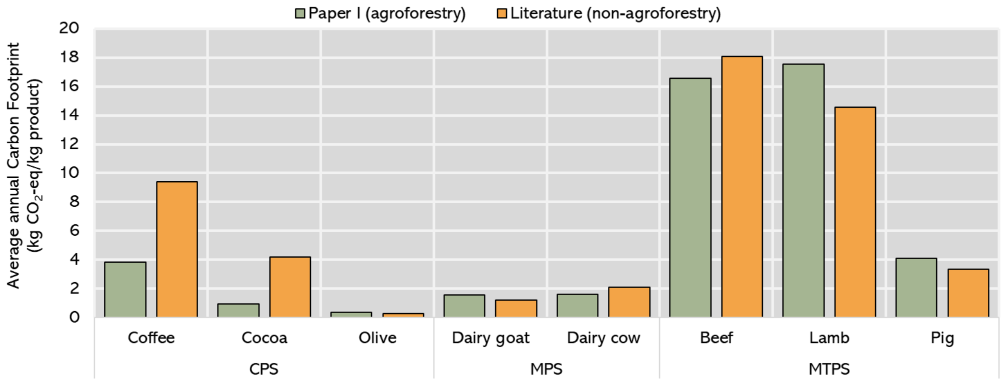


Figure 4-4. Summary of the average Carbon Footprint (CF) for three contrasting systems: Crop production system (CPS), milk production system (MPS), and meat production system (MTPS), as defined in **Paper I** (in green) and the non-agroforestry literature (average of Figure 4-1, 4-2, and 4-3 in orange). Units in kg CO_2-eq/kg product.

Figure 4-5 shows values for non-agroforestry apple ($n_{studies}$=7) and egg ($n_{studies}$= 11) production systems, including multiple observations per study (n_{obs}=13 for apples and n_{obs}=29 for eggs). Findings from **Paper II** show a CF ranging from 0.089-0.17 kg CO_2-eq/kg agroforestry apple and 0.19-1.5 kg CO_2-eq/kg agroforestry egg, which are generally lower compared to their reference systems (RS). Against the broader context of the non-agroforestry LCA literature.

The values for agroforestry apples found in **Paper II** are similar to those reported for organic apple production systems in the literature (0.07 - 0.15 kg CO_2-eq/kg apple). However, they are generally higher when compared to other production systems, such as conventional (0.014 - 0.12) kg CO_2-eq/kg apple), intensive (0.09 kg CO_2-eq/kg apple), semi-intensive (0.08 kg CO_2-eq/kg apple), or integrated systems (0.07 kg CO_2-eq/kg apple). For agroforestry egg production, the values are lower compared to other organic egg production systems in the literature (1.3-3.5 kg CO_2-eq/kg egg). Similar trends are observed in other production systems, with higher CF in intensive (3.4 kg CO_2-eq/kg egg), conventional (1.6-5.4 kg CO_2-eq/kg egg), battery cage (2.2-2.6 kg CO_2-eq/kg egg), enriched cage (2.3 kg CO_2-eq/kg egg), barn (2.7-3.5 kg CO_2-eq/kg egg), community gardens (3.5 kg CO_2-eq/kg egg), and free range and free run systems (2.4-3.4 kg CO_2-eq/kg egg).

In general, when comparing findings from **Paper II** with literature data, the agroforestry farms have a CF that is generally within a similar range to those of non-agroforestry organic systems for both apples and eggs. However, when compared to other non-AFS, such as conventional, AFS show a higher CF for apples and a lower CF for eggs. This variability is primarily due to methodological choices (discussed in **section 4.2**).

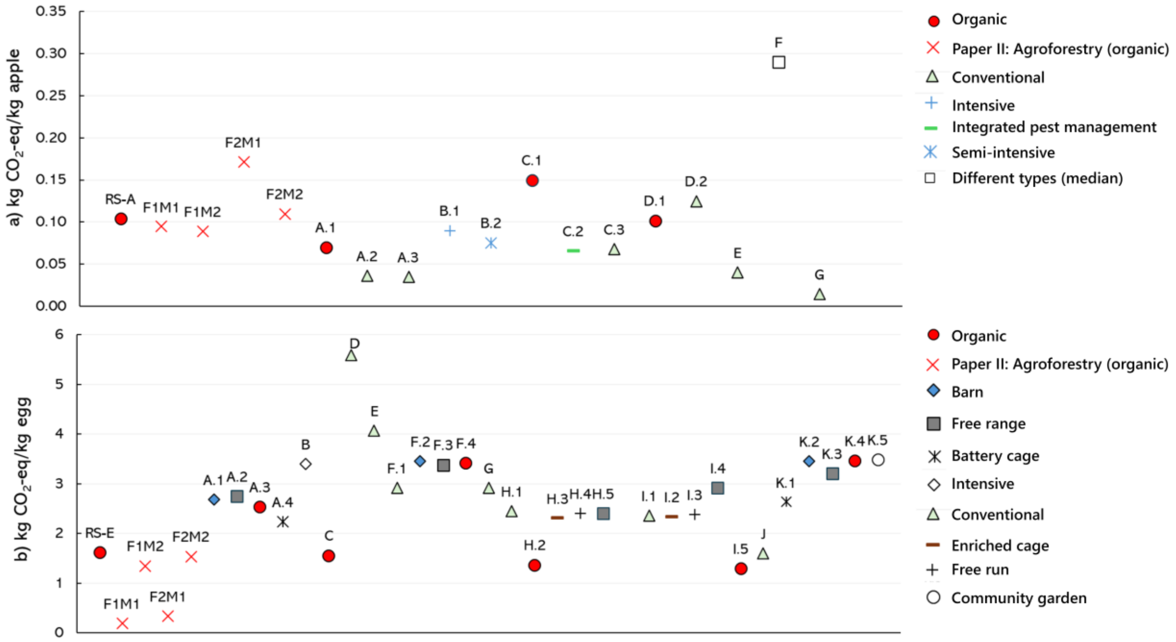


Figure 4-5. Scatter plot with literature values of non-agroforestry LCA studies from cradle-to-farm gate and different production systems in kg CO_2-eq per a) kg apple ($n_{studies}$=7) and b) kg egg ($n_{studies}$=11), including results from paper II (RS-A, RS-E, F1M1, F1M2). Data for apples: A= (Alaphilippe *et al.*, 2013); B= (Alaphilippe *et al.*, 2016), C= (Goossens *et al.*, 2017), D=(Longo *et al.*, 2017), E=(Sessa *et al.*, 2014), F=(Clune *et al.*, 2017), G=(Vinyes *et al.*, 2017). Data for eggs: A= (Dekker *et al.*, 2011); B=(Abín *et al.*, 2018), C=(Costantini *et al.*, 2020), D=(Estrada-González *et al.*, 2020), E=(Ghasempour and Ahmadi, 2016), F=(Leinonen *et al.*, 2012a), G=(Leinonen *et al.*, 2014), H=(Pelletier, 2017), I=(Turner *et al.*, 2022a), J=(van Hal *et al.*, 2019), K=(Guillaume *et al.*, 2022). A number after the alphabet value indicates multiple observations in the same study. Findings from **Paper II** are: F1M1, F2M2, F1M2, F2M2, RS-E, RS-A.

Overall, the findings across both papers (**Paper I** and **Paper II**) provide important insights into the role of AFS in the Climate Change discussion. Generally, agroforestry can have a lower CF – though in specific contexts – particularly for certain crops and animal-based products, such as coffee, cocoa, beef, and eggs. However, its mitigation potential varies depending partly on the management practices and methodological choices, as seen in the mixed results for dairy goats, lamb, pigs, and apples, where the environmental impacts are higher. Apples, for example, receive a larger share of emissions because of their higher economic value, which results in a lower CF for the eggs, a trend further influenced by yields. Also, management practices, such as compost application, feed production, and the use of pullets or spent hens, play a critical role, as some practices reduce emissions, while others increase them.

Findings in **Paper I** show that EP and AP are a relatively underrepresented category in LCAs of AFS, with only 37% and 33% of reviewed studies addressing them, respectively. Nevertheless, they still rank among the three most frequently analyzed impact categories. Findings in **Paper I** show that EP and AP are primarily reported in agrosilvicultural studies (7 out of 13 for EP and 6 out of 13 for AP), with fewer studies in agrosilvopastoral and silvopastoral contexts. This distribution reveals a significant gap in the literature, particularly for silvopastoral systems, where only 2 out of 8 studies included EP and 1 out of 8 included AP as an impact indicator. **Paper II** helps to address this gap by including EP and AP evaluations in a silvopastoral context.

Findings in **Paper II** show that, in general, EP values from agroforestry apples were 25–82% higher than those of the RS (0.28 g PO_4-eq/kg), with one farm reporting a six-fold increase. These results fall within the range reported in broader studies (0.03–3.5 g PO_4-eq/kg) (Alaphilippe *et al.*, 2013; Longo *et al.*, 2017; Zhu *et al.*, 2018). For agroforestry eggs, the opposite trend was observed, with EP values being 19–98% lower than those of the RS (34.6 g PO_4-eq/kg), with the highest values exceeding those reported in the literature (15-38 g PO_4-eq/kg) (Leinonen *et al.*, 2012b; Pelletier, 2017; Turner *et al.*, 2022b) but no more than a 1% increase in EP against the RS. For AP, values for agroforestry apples were 24-36% lower than the RS (1.3 g SO_2-eq/kg apple), but with one farm, showing more than a two-fold increase. This range aligns with the literature (e.g., 0.7–5.3 g SO_2-eq/kg apple) (Goossens *et al.*, 2017; Longo *et al.*, 2017; Zhu *et al.*, 2018). For agroforestry eggs, AP values were 17–95% lower than the RS (i.e., 44 g SO_2-eq/kg). The highest AP values (36 g SO_2-eq/kg) for agroforestry eggs remained within or below the range reported in the literature (47–91 g SO_2-eq/kg) (Leinonen *et al.*, 2012b; Pelletier, 2017; Turner *et al.*, 2022b).

In general, the findings across both papers make important contributions to understanding EP and AP in AFS, particularly by addressing the knowledge gaps in silvopastoral contexts. Together, these findings highlight trade-offs within AFS where EP and AP reductions for one product (eggs) may come at the environmental expense of another (apples). However, while AFS hold considerable potential for reducing EP and AP impacts – especially for eggs – this outcome is highly sensitive to methodological choices (discussed in **section 4.2**). Generally, EP and AP in AFS are below those in the literature, with differences largely driven by management practices, including fertilization rates for apples and feed intake for eggs.

4.1.1. Environmental interactions

The findings from **Paper I** suggest that some environmental interactions in AFS are not fully considered in LCA, which can affect the understanding of the environmental performance of AFS. Other interactions (e.g., resource loops), such as feed, manure, land, and biomass, have been possible to capture, as shown in **Paper II**, though methodologically challenging. Overall, the net environmental impact associated with different interactions within AFS remains only partially addressed in LCA. This is especially evident when contrasted with the broader environmental scope analyzed in **Paper I**. Some of these potential negative or positive farm-gate interactions with the air, water, and soil compartment in the context of a silvopastoral configuration are illustrated in **Figure 4-6**, based on findings from **Paper II** and the literature.

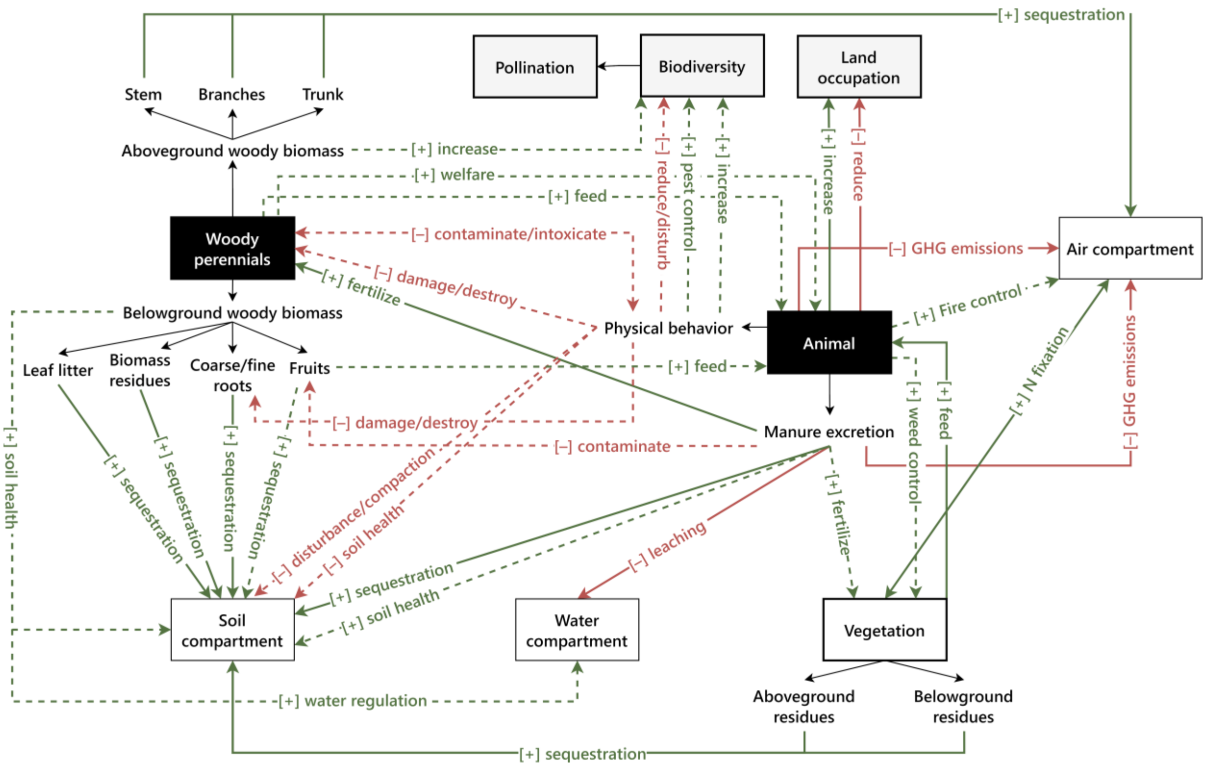


Figure 4-6. Non-exhaustive list of potential first-level interactions influencing the environmental and climate impacts of food produced in a general silvopastoral configuration at the agroforestry farm gate. Positive and negative interactions of animals integrated in a woody perennial system are shown in green and red, respectively. Solid lines represent interactions included in **Paper II**. The dotted lines represent potential first-level interactions identified in the general agroforestry literature that require further investigation for a more comprehensive assessment of the net effects.

Integrating domestic animals into woody perennial systems introduces new complex dynamics at the farm level that are challenging to capture in LCA and are worth questioning during the assessment. This complexity is further influenced by factors like local conditions (climate, soil type), management practices (conventional, extensive, organic), and other aspects (animal breed, tree species), which can shape the direction and magnitude of the environmental and climate impacts of AFS as found in **Paper I**.

Some interactions, such as animal activity, particularly pecking, scratching, and trampling, can affect negatively the soil compartment (e.g., reduce soil health) (Bosshardt *et al.*, 2022). Animals can compact the soil and destroy or damage the root zones, which can limit the growth of trees (Sales-Baptista *et al.*, 2016). These negative effects have been observed in studies combining poultry with apple orchards in the Netherlands (Bestman, 2017). Though barriers can be used to protect young trees from animals, this can create an environment for pests (e.g., mice). Nevertheless, biomass from trees, manure deposition, and herbaceous vegetation can impact positively the soil compartment by increasing soil health (e.g., soil fertility) (Reed *et al.*, 2017), which can in turn benefit the air compartment by enhancing C-seq and reducing GHG emissions (Feliciano *et al.*, 2018). These interactions are partly addressed in **Paper II**, which, despite variability, shows that models for estimating C-seq can be applied in LCAs (discussed in **section 4.2.1**). In **Paper I**, 15 of the 32 selected studies included C-seq aspects in their LCA, although only two incorporated data on soil health. Soil health models like LANCA could be refined and incorporated to better assess AFS in LCAs (De Laurentiis *et al.*, 2019). However, there is still no consensus on the most effective way to integrate these factors into the LCA framework.

The presence of animals can help mitigate fire risks by providing natural vegetation control, benefiting the air compartment (Sollen-Norrlin *et al.*, 2020). However, they can also increase overall GHG emissions, particularly due to feed production, CH_4 emissions, and manure-related processes such as NH_3 volatilization (**Paper II**). These factors can complicate the anticipated environmental benefits of agroforestry at the whole-farm level. As shown in **Paper II**, total farm emissions can almost double when animals are introduced at high stocking densities, even if apple yields remain similar.

This raises the question of how introducing woody perennials into land already supporting animals might impact whole-farm mitigation efforts. This approach could theoretically offer a better solution by creating additional space for trees and lowering animal densities. Regardless of whether animals are integrated into woody perennial systems or woody perennials are introduced into existing grazing land, the effects of GHG emissions on the air compartment can be captured using well-established impact assessment models for calculating the CF as described in **Paper II**. The net effect can be then estimated by incorporating the C-seq potential. However, as shown in **Paper I** and **Section 4.2.1**, it can introduce significant variability.

Careful attention to N requirements should be placed, particularly in soil with low retention capacities (e.g., sandy soil). Without aligning N inputs with current fertilization rates and actual crop and soil requirements, manure deposition may exacerbate N losses negatively impacting the water compartment (Leip *et al.*, 2019). In such cases, attempts to enhance C-seq may inadvertently increase the risk of leaching and N_2O and CH_4 emissions (Dixon, 1995; Alemu, 2014). The environmental burdens of manure have been challenging to define in **Paper II**, where it was assumed that it was not wasted or applied in excess of crop nutrient needs. This assumption was based on the observed relatively low stocking densities and large outdoor area, which may not be the case for other AFS. Thus, the role of manure could be further modeled more critically. For example, the theoretical approach proposed by Leip *et al.* (2019) to define nutrient requirements could be used in future research.

The water compartment can be positively impacted, as soils enriched with organic matter from manure and biomass deposition have an improved water retention capacity, reducing runoff (Anderson and Sinclair, 1993; Atangana *et al.*, 2014b). **Paper II** partially addresses some of these aspects (e.g., leaching). Despite the advancements in LCA regarding water footprint methodologies (Berger and Finkbeiner, 2010; Gerbens-Leenes *et al.*, 2021), **Paper I** shows that only six out of the 32 studies considered water-related factors like resource depletion, water footprint, or scarcity. However, none of these methods currently account for how water regulation services (e.g., retention and infiltration) interact in AFS.

Trees can increase animal welfare and supply forage for animals through browsing or fallen fruits, while herbaceous vegetation offers accessible feed during grazing periods (Torres *et al.*, 2020; Lamnatou *et al.*, 2022). Nevertheless, there could be a risk of animal intoxication regardless if AFS are conventionally or organically managed (De Vries *et al.*, 2006) given that sometimes chemical applications are needed in orchards or fields, which are prohibited in egg or other types of production systems. Similarly, if manure is deposited near orchards, it can lead to fruit contamination (Theofel *et al.*, 2020). Animals in AFS have also been reported to enhance biodiversity (birds and vascular plants) (e.g., in the Portuguese *Montado*) (Simonson *et al.*, 2018). They can also contribute to weed and pest control through insect consumption, as observed in chicken-orchard pasture systems in the Netherlands (Bestman, 2017) and in a global meta-analysis (Pumariño *et al.*, 2015). However, foraging behavior in AFS has also been reported to disturb or reduce the populations of auxiliary fauna (spiders, earthworms) (Bosshardt *et al.*, 2022) and soil lichens and mosses (Concostrina-Zubiri *et al.*, 2017). Similarly, intensive grazing and high animal densities in AFS have been associated with the decline of natural cork oak regeneration due to the consumption of seedlings and saplings (Arosa *et al.*, 2017). **Paper I** shows that only five studies reported on biodiversity impacts which were most often measured at the endpoint level. While some characterization factors may be used as proxy measures for biodiversity in Europe (Knudsen *et al.*, 2017), there is a lack of applicable models specific to agroforestry that cover all biodiversity levels and go beyond species richness. The supplemental forage could reduce the need for imported concentrates, positively impacting the air compartment by lowering GHG emissions (Lamnatou *et al.*, 2022). **Paper II** partially addresses some of these issues. Pest control can be partially reflected through reduced chemical use and lowered EP, and contamination or intoxication effects can be reflected by reduced yields or animal populations. However, the extent and magnitude are debatable, requiring further field experiments. For feed intake, the approach in **Paper II** remains theoretical and requires refinement, highlighting a need for experimental data on animal foraging behavior specific to the agroforestry case study under analysis. The benefits of foraging may be reduced if the supply of concentrate is not adjusted accordingly.

Finally, agroforestry can maximize land productivity, allowing for the simultaneous production of food and other products as shown in **Paper II** (Dodds *et al.*, 2019). However, while the total LO can decrease at the farm level, reducing land for one product can come at the expense of another. This issue was captured in **Paper II** (M1), where it was found that LO decreased by 10% for eggs due to savings in farm area from the outdoor run, but land required for apple production rose by 4-42% because of the feed production for the eggs used during the rearing and production phase. Some agroforests can drive land expansion, risking encroachment on primary forests, as documented in cocoa agroforestry plantations in Brazil (Rolim and Chiarello, 2004; Ollinaho and Kröger, 2021). This interaction can be incorporated in LCA by estimating direct and indirect land use changes (De Rosa, 2018), though this remains a challenging aspect to measure accurately.

4.1.2. Importance of post-farm gate impacts

While the above-mentioned interactions primarily occur at the 'agroforestry farm gate' level, the environmental impacts of agroforestry food products also extend beyond this stage (i.e., 'agroforestry non-farm gate'). For example, in MTPS operating in the *Montado* (**Paper III**) cattle and pigs may benefit briefly from the traditional agroforestry ecosystem before or after transitioning to intensive and conventional systems with typically greater environmental costs in terms of feed and land use (Horizonte de Projecto, 2017). However, the majority of LCAs reviewed in **Paper I** do not include those interactions. These extended impacts and additional costs or benefits should also be added to the overall equation to determine the net environmental and climate effects. For instance, some might highlight biodiversity gains in AFS but overlook potential declines in beneficial species linked to the fattening period for cattle or the production period of pigs.

Post-harvest processes can contribute to environmental impacts (particularly GHG emissions) though the degree varies across food products (**Paper II**). For apples, post-harvest emissions are due to the energy and diesel used during storage and transport which may vary depending on the efficiency. In some fruit chains, these emissions (including manufacturing) may exceed those from production, as seen in chocolate production (Recanati *et al.*, 2018; Pérez-Neira *et al.*, 2020). In **Paper II**, post-harvest emissions can contribute 33-49% and 13-57% of the total CF for apples and eggs, respectively (with the higher range in M1 for eggs and M2 for apples). AFS can benefit from adopting organic principles, as localized supply chains allow direct farm sales or limited distribution, reducing transportation distances, as shown in **Paper II** (Ušča and Aļeksējeva, 2023).

In summary, when addressing the first part of the central research question, *what insights can LCA provide when comparing the environmental and climate impacts of different food products from AFS?*, the findings across both papers show that food produced in AFS generally performs better than those produced in non-AFS at the agroforestry farm-gate level, with crop and milk production systems achieving better outcomes per kg product compared to meat production systems. However, the environmental advantages appear to be more marginal than the general agroforestry literature suggests. Also, shorter and local value chains can benefit the CF of food products in organic AFS. Overall, results are context-dependent and influenced by methodological approaches (**section 4.2**). Thus, a more nuanced approach is needed when addressing this question as only part of the complex positive and negative environmental interactions occurring in the agroforestry food value chain are captured or included in LCA.

4.2. Methodological challenges

In addressing the second part of the central research question '*How do different methodological choices influence the outcomes of these assessments?*', findings in **Paper I** and **Paper II** show that variations are driven by several factors beyond differences in management practices. For example, default emission factors can lead to high uncertainty compared to site-specific or localized factors. Variations in the functional units, allocation approaches, and the handling of multifunctionality can yield different results. Advancements are also necessary for inventory data related to organic feed and compost mixes. The lack of representativity of the background databases such as using average or industry values for upstream emissions (e.g., nursery and establishment activities or organic feed production), may not accurately reflect the local environmental conditions of the modeled agroforestry practices.

4.2.1. Carbon sequestration

Complementing the above-mentioned methodological challenges, results in **Paper I** indicate a significant gap in the LCA literature of AFS where many studies do not fully consider C-seq in the CF. While the reviewed LCA studies in **Paper I** frequently emphasize agroforestry as an important mitigation pathway for climate change, only a handful were found to account for C-seq in the woody biomass or soil (Quevedo-Cascante *et al.*, 2023). Rarely were both included. Findings in **Paper I** show that some studies treat soil and biomass C as neutral, assuming no impact on climate. In other words, some studies assume that soil C inputs and outputs balance out and that C stored in woody biomass eventually returns to the atmosphere. This approach stems partly from practical simplifications and methodological challenges involved in accurately measuring C-seq over time. This is because C dynamics are often temporary as disturbances (e.g., tilling, burning) can release stored C and other factors (e.g., soil type, climate, crop type, tree age, tree density, and management practices) can introduce variability (Goglio *et al.*, 2015; Cardinael *et al.*, 2018). However, other scholars argue that when a system actively sequesters C and holds it outside the atmosphere for an extended period (e.g., 20 years), it offers measurable climate benefits (De Stefano and Jacobson, 2018), especially under practices that enhance organic matter (e.g., agroforestry, organic) (Brandão *et al.*, 2013).

Various methods are available to determine the C-seq potential in the woody biomass, each differing in how they handle C storage duration and release (**Figure 4-7**). **Paper II** applied the most conservative method, developed by Clift and Brandao (2008), which assumes that C storage is temporary and that the stored C will eventually be released back into the atmosphere (due to decay or other ecological processes). Consequently, the amount of C credit diminishes as well so it only gives partial credit for C-seq each year. In other words, C stored is treated as a short-term benefit that delays emissions (i.e., slows down the contribution of GHG to global warming), but the method assumes much of it will be released (i.e., storage is not permanent). Thus, the factors used for the estimations represent a time-based decline in C storage potential. This method states that storing 1 t CO_2 for 1 year is equal to avoiding around 8 kg of CO_2 emissions in a 100-year perspective.

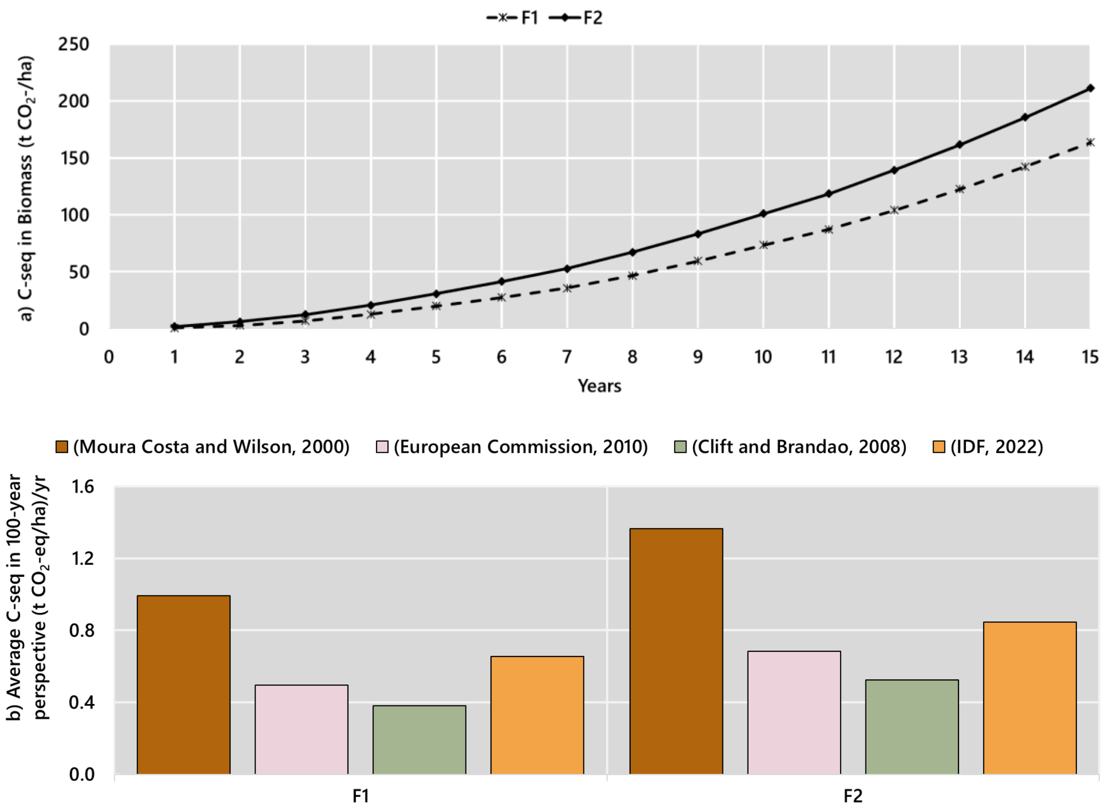


Figure 4-7. a) Carbon sequestration (C-seq) potential in the woody aboveground biomass over one crop rotation (15-year period) for F1 (straight line) and F2 (dotted line) and b) different methods for estimating the average CO_2-eq sequestered in woody AGB per hectare per year for F1 and F2

Other methods provide different estimates for C-seq in the woody biomass. For example, the Moura Costa and Wilson (2000) method gives a 2.6-fold increase compared to Clift and Brandao (2008) (**Figure 4-7**). This method is more optimistic because it does not consider the fate of C after the storage period, giving a constant factor for the temporary C stored in the biomass (0.02 t CO_2-eq/yr). This means that storing 1 t of CO_2 for one year avoids the impact of 20 kg of CO_2 emissions in 100-year perspective.

The ILCD method (European Commission *et al.*, 2010) uses a different factor to adjust the climate credit for C-seq (-0.01 kg CO_2-eq/kg CO_2 year stored). Similar to Clift and

Brandao (2008), more credit is given to C stored earlier in the rotation because it remains stored for a longer period (so it provides a greater climate benefit) and less credit is given to C stored later in the cycle. In other words, it takes into consideration the number of years the emission is delayed. This method provides a partial climate benefit for short-term storage (e.g., 15 years), with full credit only given if C remains stored for the full 100-year horizon (e.g., 15 years = 15% of the credit). In other words, each year that 1 t CO_2 is stored equates to a reduction of 10 kg in CO_2 emissions in a 100-year perspective. This method gives a 1.3-fold increase compared to Clift and Brandao (2008) (**Figure 4-7**).

The IDF (2015) method adjusts the ILCD factor according to the designated responsibility window (e.g., the rotation period) based on value choices. Thus, unlike ILCD which provides the full climate benefit of C storage only if it is stored for 100 years, this method provides the full climate benefit (100%) over the responsibility window (e.g., 15 years). So, 15% of the credit is given and a larger factor is used (i.e., -0.06 kg CO_2-eq/kg CO_2 year stored). For longer timeframe storage (e.g., 100 years), a smaller factor is used (i.e., -0.01 kg CO_2-eq/kg CO_2 year stored). However, this factor is not multiplied by the number of years the emission is delayed. This method gives a 1.7-fold increase compared to Clift and Brandao (2008) (**Figure 4-7**).

Regarding soil C-seq, the method by Petersen *et al.* (2013) applied in **Paper II** provides reliable estimates for temperate European climates like Austria, making it more specific than the IPCC's Tier 1 factors (Goglio *et al.*, 2015). However, it may be less applicable in warmer regions like Portugal and the method requires estimates of the contributions from both above- and below-ground crop residues, where limited data is available. Nevertheless, unlike other models that use generic inputs (see models in Goglio *et al.*, 2015), Petersen *et al.* (2013) use a dynamic approach, meaning that it captures both the cumulative impact over long periods and how C is added and released in response to management and natural processes.

In summary, findings from **Paper II** highlight the significant variability in estimating C-seq. When focusing on the C-seq in the woody biomass alone, differences are up to 2.6-fold depending on the method. These overall variations in C-seq are aligned with the findings in **Paper I**, which show, for example, that the high variability in MTPS, (with negative values among silvopastoral and agrosilvopastoral groups, especially for beef and calf), is driven mainly because the different C-seq approaches. This underscores the need for more standardized methodological choices when assessing the net CF of AFS.

4.2.2. Multifunctionality and N pathways

An important aspect of LCA is the handling of multifunctional processes, as it significantly influences outcomes. ISO standards suggest avoiding allocation, ideally by partitioning the system into sub-processes (ISO, 2006a). When this isn't feasible, allocation should be based on the relationships between products, like economic factors. In **Paper II**, the latter approach was used in M1 and the former in M2 and model 3 (M3). The strength of M1 lies in its ability to account for interconnected resource loops (e.g., manure, land, feed, biomass) that are difficult to separate. The disadvantage is that it can disproportionately attribute emissions to high-value products like apples, which may lead to biased results favoring secondary activities that are relatively more environmentally burdensome but with low economic value, such as egg production. Alternatively, M2/M3 linked emissions directly to subsystems and their specific products. Here, high-emission activities (e.g., feed production) were not tied to the economic value or yield of the main product (apples). However, separating and modeling resource loops independently is challenging, especially with compost containing manure (for the egg subsystem) and plant biomass (for the apple subsystem), as well as accurately tracking quantities used or exported in the orchard. This challenge is illustrated in **Figure 4-8** and **Figure 4-9**, where the distribution of N flows associated with manure excretion and compost has been subdivided according to the subsystem it theoretically belongs to (biomass allocated to the apple subsystem and manure to the egg subsystem).

Findings from **Paper II** show that N is particularly challenging to track in an isolated manner because it originates from multiple sources and undergoes various transformations. Biological interactions, such as microbial activity in composting add further uncertainty to N tracking in M2 and M3 in **Paper II**. These transformations mean that a portion of N may never reach the field in its intended form, and precise attribution to a specific subsystem becomes difficult (Boldrin *et al.*, 2009). This is because manure, which undergoes biological changes during composting, alters the composition and bioavailability of N when applied to the apple subsystem (Eusufzai *et al.*, 2013). This means that the N associated with manure applied to the agroforestry land is not identical to the N excreted initially by the hens. Furthermore, isolating the contribution of N from manure applied directly to the fields for apple trees is particularly difficult. As N enters the system, a portion of it is potentially retained in the soil and taken by the trees. M1 can capture these interactions that are difficult to partition in M2 and M3 by using economic allocation. These varied N pathways make accurate quantification and allocation complex where the dynamics between storage, handling, and eventual application blur these boundaries. These difficulties in accurately quantifying N flows highlight broader methodological challenges for LCA modeling of AFS. The complexity of biological interactions, especially in composting and manure management, complicates the use of subdivision for handling multifunctionality in AFS. To improve accuracy, more advanced N tracking methodologies are necessary.

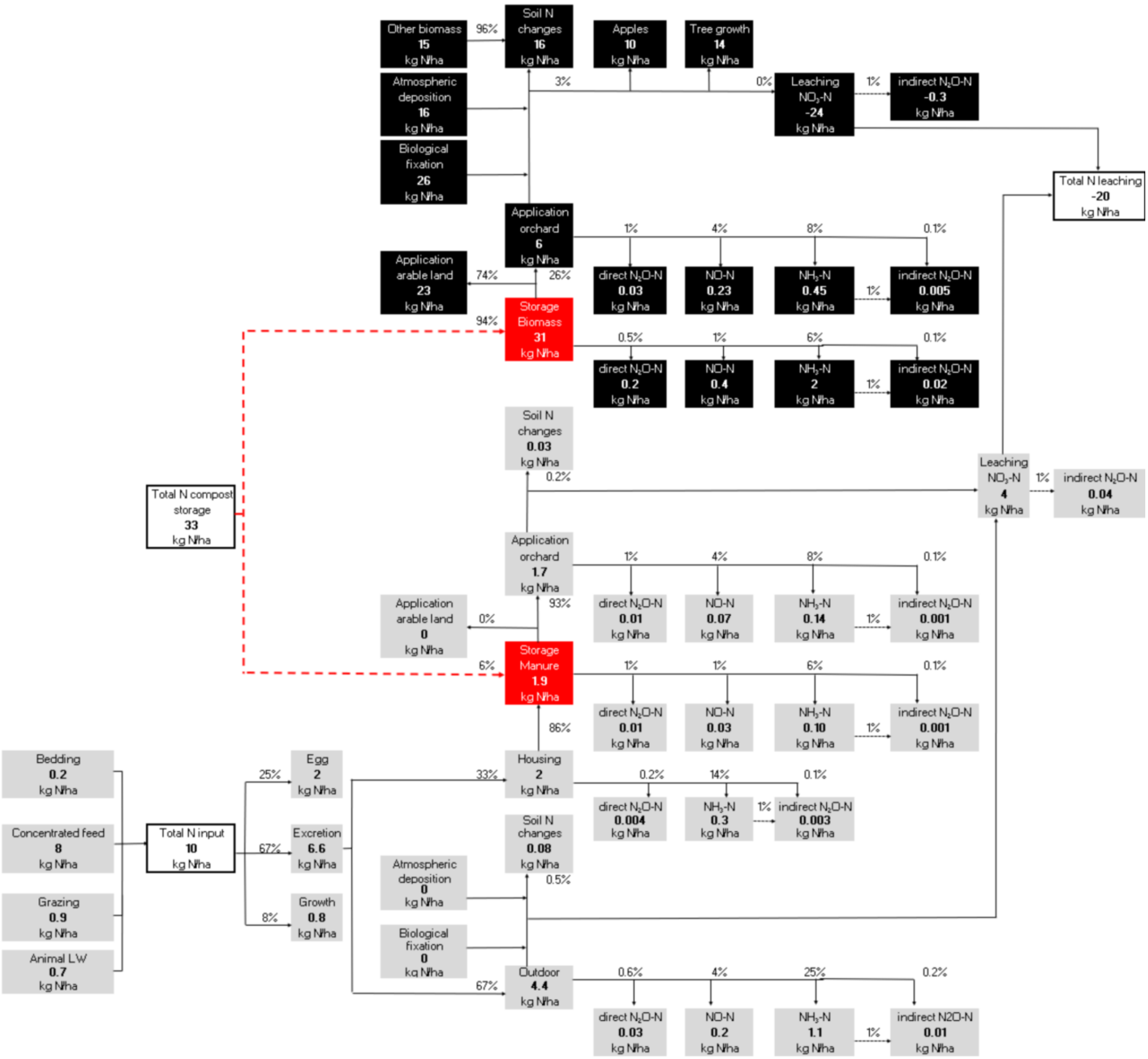


Figure 4-8. Distribution of N flows associated with manure excretion and compost for F1. Black boxes= apple subsystem; Grey boxes= egg subsystem; White boxes= total N. In red= the stored compost mixture, which has been subdivided according to the subsystem it belongs to (biomass allocated to the apple subsystem and manure to the egg subsystem).

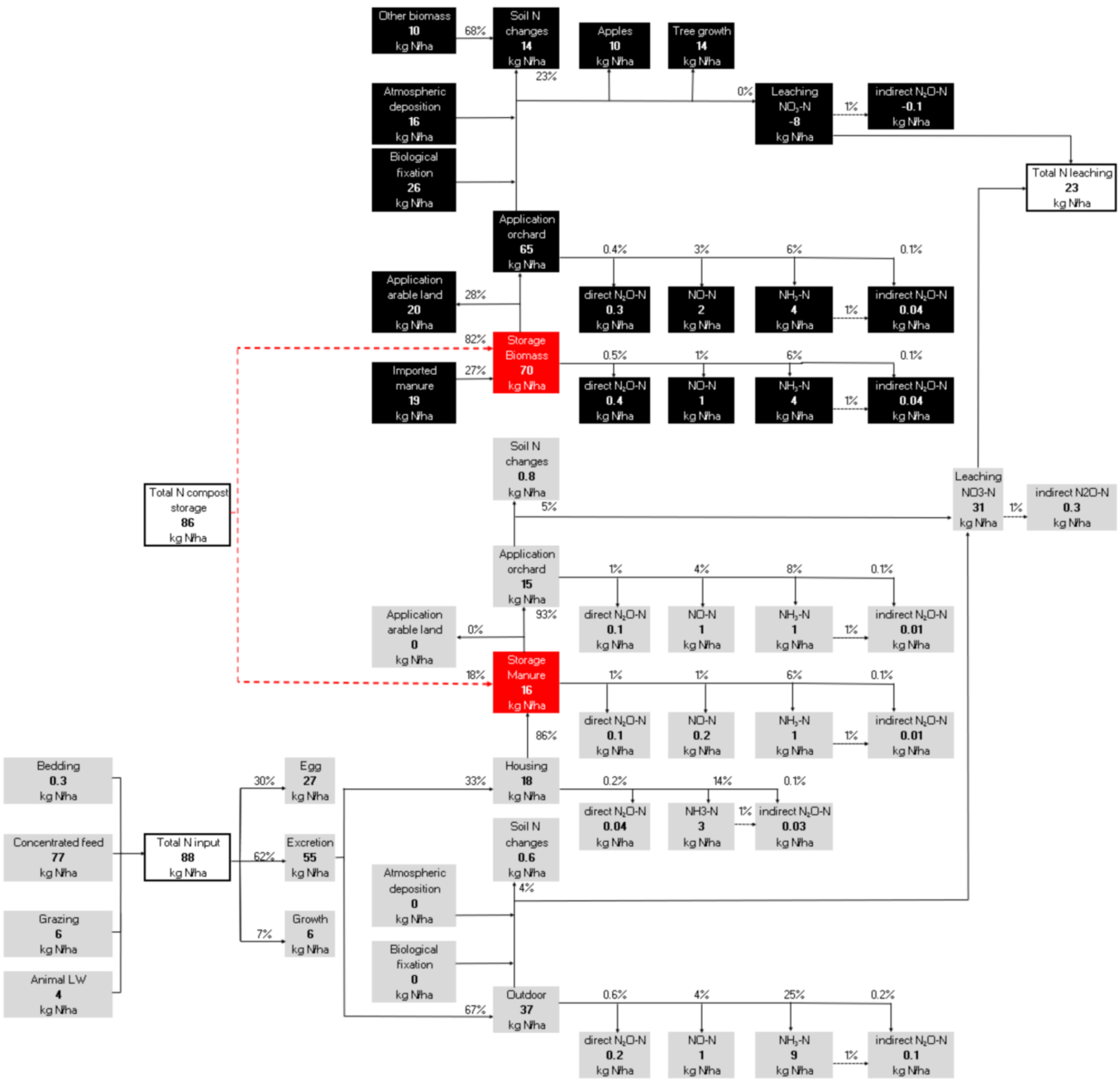


Figure 4-9. Distribution of N associated with manure excretion and compost flows for F2. Black boxes= apple subsystem; Grey boxes= egg subsystem; White boxes= total N. In red= the stored compost mixture, which has been subdivided according to the subsystem it belongs to (biomass and imported manure allocated to the apple subsystem and manure to the egg subsystem).

N leaching can vary significantly, ranging from 5% to 50% of the total N inputs in the literature (Sainju, 2017). The IPCC provides a default leaching and runoff factor for N that is lost. This factor is typically 0.3, which means that 30% of the applied N (whether from synthetic fertilizers, manure, crop residues, or other sources) is assumed to be lost to the environment via leaching and runoff (IPCC, 2019). However, this default factor may not accurately reflect local conditions, as variables such as soil type, climate, and agricultural management practices can influence the N pathways (Brentrup *et al.*, 2000). To estimate more accurately N turnover and its effect on EP, a gross N balance approach was used in **Paper II**. Although this method introduces uncertainties due to the need to estimate multiple parameters, it was coupled with country- and farm-specific data when available, making it more robust. If the default IPCC values had been used, the estimated leaching for F2 would have been approximately 1.9 times higher, and for F1, it would have been overestimated (as the N balance in F1 showed no N surplus, indicating no theoretical risk of leaching). These differences highlight the importance of using site-specific data and accounting for factors such as changes in the soil N pool and tree growth, which significantly reduces the estimated leaching compared to generalized approaches. This highlights the need for adapting methodologies to take into consideration specific aspects relevant to AFS.

Overall, depending on the modeling approach, findings from **Paper II** suggest some cross-sectional challenges. For example, potential N leaching is the most significant contribution to the EP impact category but also contributes to the CF through indirect N_2O emissions. Overall, although the models tested in **Paper II** are not exhaustive of all factors influencing LCA results, the approach makes positive advancements by applying recent recommendations, such as the AR6 characterization factors, and using species-specific and dynamic models for C-seq in the soil and the woody biomass, providing also more accurate insights into soil C contributions and exported biomass impacts. Moreover, by using empirical farm data and adopting a gross N balance approach, nutrient flows and possible contributions to different subsystems are estimated more accurately, though some challenges remain.

In summary, in addressing the second part of the central research question, *how do different methodological choices influence the outcomes of these assessments?,* findings in both papers show that LCAs of AFS are highly sensitive to modeling approaches and influenced by the various methodological choices which can partly shift interpretations. The reliance on default emissions factors instead of site-specific data can also result in significantly higher values. Variations in C-seq methods, even when using similar functional units and system boundaries, can lead to considerable differences. Tracking nutrient pathways, particularly N, poses additional cross-sectional challenges in multifunctional systems.

4.3. Limitations

4.3.1. Research framework

Although LCA may be well suited for assessing linear production systems (i.e., industries with a clear distinction between the technosphere and ecosphere), it may not fully capture the resource loops in biological production systems (Notarnicola *et al.*, 2017). For example, the complex interactions between trees, crops, and animals in AFS (e.g., manure flows within and outside the system boundaries, as shown in **Paper II**). While LCA is a good tool for quantifying emissions and resource use, it can be difficult to use for capturing complex biogeochemical or hydrological processes and other long-term ecological dynamics that change over time (e.g., biodiversity, pest and disease control, as discussed in **Paper I**). This is because LCA typically provides a snapshot rather than a dynamic assessment, which can overlook temporal variations from a changing environment (e.g., biophysical and seasonal changes, including management activities and species behavior) (Hauschild *et al.*, 2018). Furthermore, the focus on the environmental matrix alone can miss broader and interconnected issues, such as the socio-economic and public health ramifications.

It is difficult to account for broader landscape-level effects because the system boundary in LCAs typically focuses on the value chain of a specific product. This means that wider ecological interactions, such as habitat connectivity, water flow regulation, or biodiversity corridors, are often excluded from the analysis (Notarnicola *et al.*, 2017). Thus, positive landscape-level impacts (e.g., erosion control or regional carbon storage) and cross-farm mixedness interactions (e.g., resource loops and exchange), may not be fully captured (Oomen *et al.*, 1998). At the same time, potential negative landscape-level impacts (e.g., competition for water resources or unintended consequences of agroforestry expansion) may not be fully reflected within product-focused boundaries (Rao *et al.*, 1997). Thus, it is challenging to assess both positive and negative effects beyond the immediate production area, leading to an incomplete picture of agroforestry's overall impact on a larger scale.

LCA is typically applied at the relative level, which may not reveal the full scale of environmental impacts (Gerten and Kummu, 2021). For example, while agroforestry might perform better than non-AFS in some environmental indicators, as shown in **Paper II**, relative LCA doesn't show whether these impacts are within the planetary boundaries or if they are 'good enough' (Hauschild, 2015). Given that food products often come with negative environmental impacts, any benefits must be justified. An absolute LCA could provide a clearer understanding of whether AFS providing the same function (e.g., 1 kg of protein), in their various configurations (e.g., silvopastoral vs. agrosilvicultural), are meaningful on a larger scale. However, the methodology is still new and under development.

Despite these limitations, LCA remains a valuable tool for identifying key environmental issues in AFS at the product level. The relative approach enables a comparative analysis of similar products, making it suitable for identifying trade-offs and hotspots associated with certain management choices and also across the food value chain. However, complementing LCA with other methodologies and models is necessary to capture the full range of impacts and interactions in such systems.

4.3.2. Research design

The use of case studies can limit the generalizability of the findings because the assessments take place in a real-world setting, where variables cannot be controlled, and the pool of farms analyzed in **Paper II** and **Paper III** was small. The selected UAOs may introduce bias, as they could represent better-performing systems compared to those excluded or not included at all within the MIXED network. The same bias in the scientific literature has been discussed for **Paper I**. Furthermore, the different methodological approaches used in each paper also present challenges for cross-case comparisons, as the varying depth and focus of analysis across papers were different. Nevertheless, the inclusion of multiple agroforestry configurations in **Paper I** further broadens the understanding beyond European contexts, helping to mitigate concerns about generalizability. Also, the multiple-case study design draws connections between these approaches, allowing for meaningful comparisons and the identification of patterns and trends across the different AFS and the literature.

Specific environmental impact categories (i.e., CC, EP, AP, and LO) were assessed in **Paper II**, while other important indicators identified in **Paper I**, were not explored. The decision to focus on these particular categories was influenced by the exploratory nature of the thesis and practical constraints. Models for some impact categories, like biodiversity, are still in the early stages of development within the LCA framework (Turner *et al.*, 2019), and incorporating them would have required the development of specific methods, which was outside the scope of this thesis. The selected categories are widely recognized in the food LCA and agroforestry literature and are better developed in terms of LCIA methods (Notarnicola *et al.*, 2017). This allowed for a more robust assessment, as well as comparisons with existing studies in the literature.

Finally, the use of an ALCA approach does not account for broader market-driven or system-wide changes as typically captured in a consequential LCA. This could limit the exploration of important indirect environmental impacts or rebound effects, which might arise in a real-world application of AFS on a larger scale (e.g., deforestation through indirect land use changes, carbon opportunity costs) (Schmidt, 2008; Prox and Curran, 2017; Weidema *et al.*, 2018). Despite the above-mentioned limitations, ALCA is widely used in agri-food research because it captures the immediate impacts of production systems without introducing confounding and uncertain variables from external market forces. Thus, the findings provide a focused analysis that can be useful for identifying the specific factors that contribute to environmental impacts, such as the choice of tree species and management practices.

4.4. Perspectives and Outlook

While this thesis provides insights regarding the environmental synergies and trade-offs of AFS from a supply-side and product-level perspective, other areas remain open for further exploration. For example, a key question for the future of well-managed AFS is its scalability within the existing food system, especially as global population projections suggest a need to increase food production by 70% by 2050 to meet urbanized, wealthier demands (FAO, 2009). This is a challenge for AFS, which generally requires more land than 'business-as-usual'. For example, a silvopastoral system that integrates multiple animal species and nut-bearing trees in rotational pastures requires approximately 2.5 times the land area relative to a single-commodity production system (Rowntree *et al.*, 2020).

A dominant school of thought suggests that technological advancements could allow AFS to meet market-driven demands with less environmental impact (Garnett, 2014). In other words, eco-efficiency and productivity are sufficient to address environmental challenges without a need to fundamentally alter production practices, market structures, or consumption patterns (Pelenc, 2015). However, while technological advances can increase eco-efficiency, they risk inducing a rebound effect (i.e., as efficiency rises, so does consumption) (Onat *et al.*, 2017). This effect, often unaccounted for in ALCA, highlights the limitations of relying solely on productivity, as efficiency gains could be offset by higher demand and limited by biological constraints (Prox and Curran, 2017). A less dominant school of thought in agricultural research suggests that transformation efforts should align AFS with 'genuine' needs rather than market-driven demands (Garnett, 2014). An example of such needs can be addressed on the demand side of the value chain through consumption patterns (e.g., nutritional and dietary requirements). Yet the complexity of global food markets, where consumption in one place often depends on production in another, creates shared environmental impacts and responsibilities between the Global North and the Global South. This requires identifying places where high-impact foods (e.g., meat) could be downscaled without exacerbating nutritional gaps and assessing the context in which AFS might fulfill regional needs. Acknowledging this fundamental issue is critical to addressing the oversupply of food products driven by affluent and unhealthy consumption patterns (Rockström *et al.*, 2020; Sun *et al.*, 2022). This raises the question: What is the scalability potential of well-managed AFS to meet contextual nutritional needs and planetary health goals under sufficiency principles (Brinken *et al.*, 2022) and across different configurations (silvopastoral, agrosilvicultural, agrosilvopastoral)? What defines a well-managed AFS, and what strategies or criteria can be established to authenticate genuine adherence to responsible environmental practices? Which agroforestry configuration should be further promoted and under what conditions do they offer the most effective path for health and environmental and climate improvement? Or as Shackleton *et al.*, (2016) have previously questioned, could reducing negative interactions yield more statistically significant environmental benefits than increasing positive ones? And could efforts to enhance positive ones inadvertently and significantly intensify negative interactions? Is LCA the right tool for answering these questions? Exploring these fundamental issues for the viability and meaningfulness of AFS is important.

Other relevant issues, such as bureaucratic burdens, costs, and land tenure, are barriers to scaling (Tsonkova *et al.*, 2018; Teixeira *et al.*, 2019). Farmers on rented land may avoid AFS due to the long-term nature of tree planting, which doesn't align with short-term leases (Mugure *et al.*, 2013; Persha *et al.*, 2015). The presence of trees may decrease land value, and without clear financial incentives, landowners may eventually cut them down (Mugure *et al.*, 2013; Persha *et al.*, 2015). Uncertainty in the political landscape, including the lack of support, could further complicate the widespread adoption and scale of AFS (Tsonkova *et al.*, 2018; Sollen-Norrlin *et al.*, 2020; Thiesmeier and Zander, 2023). This opens the question of whether AFS are a near-sighted solution. And how can policy incentives be structured to make AFS both more financially viable and conservation-focused, even when they may result in lower yields? Under what conditions do lower yields (or downscaling production) align with broader planetary and public health goals? And what is the ideal policy landscape for supporting the long-term adoption of well-managed agroforestry configurations while balancing economic incentives and land tenure constraints? Expanding the assessment and including these issues could provide a clearer understanding of profitability, adoption, and trade-offs beyond the environmental matrix.

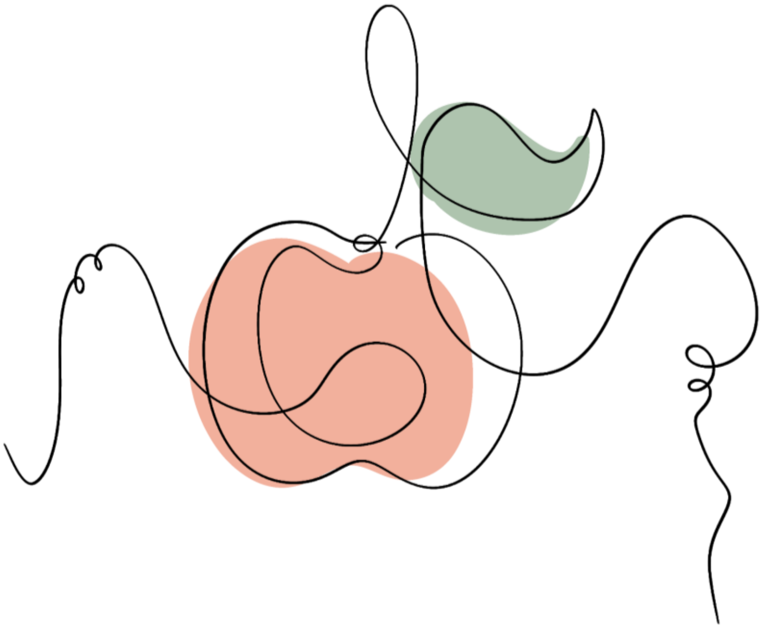
In summary, mitigation efforts in the current food system arguably require more than merely planting trees or integrating animals into perennial systems. Given that AFS remains embedded in a global food system driven by growth, oversupply, and excess, research efforts could consider shifting from an eco-efficient paradigm (beyond agronomic productivity and incremental changes), to rethinking both production and consumption patterns more broadly, where environmental issues are not exclusively determined by market-driven forces. This shift could orient agroforestry and LCA research, policy-making, and research funds toward a paradigm that acknowledges ecological limits, sufficiency, public health, agroecological principles, and food security. Just as LCA could integrate with other methodological approaches (e.g., biodiversity models) to support decision-making, agroforestry research could combine with high-impact actions (e.g., downscaling ruminant populations) to achieve greater environmental impact reductions and provide a more comprehensive analysis of its viability, scalability, and meaningfulness.

4.5. Recommendations

Specific methodological recommendations drawn from the multiple-case study comparisons include using biomass estimation approaches tailored to local conditions (e.g., climate, soil type, tree species, and age) and models that account for the temporal dynamics (e.g., release and delay) of C-seq. Also, a clear differentiation of C-seq sources, such as manure deposition and biomass, is essential for identifying which subsystems contribute most significantly to C-seq, enabling accurate allocation of potential C-seq benefits and preventing an 'over-assignation' between products. Additionally, in terms of modeling approaches, it's important to recognize that the choice of modeling lens can introduce bias by selecting a model that benefits the most the farm or product. Transparency about the modeling criteria and sensitivity analysis is therefore crucial. Adapting N balances to country-specific data is also essential for accurate evaluations. In this context, high-quality data on feed intake and feed composition is particularly relevant, as it affects both excretion estimates and, therefore, the EP and CF. Moreover, although LCA has faced criticism for excluding important beneficial dynamics in certain agri-food systems (e.g., organic, low-input) (Othoniel et al., 2016), focusing solely on, for example, ecosystem services may inadvertently introduce a positive bias as well. Thus, the scope of assessments should also acknowledge issues within and beyond the agroforestry farm gate, integrating both positive and negative interactions to achieve a more balanced and context-sensitive assessment, as discussed in **section 4.1.1**.

Promoting better management practices is also important. Recommendations include prioritizing the integration of trees into existing animal production systems. Otherwise, integrating existing animal populations (as opposed to new populations) into plantations. Also, minimizing soil disturbance through techniques like grafting in orchards and recycling organic inputs within the agroforestry boundary, such as pruning residues and manure. Management practices should be tailored to crop-specific needs, optimizing fertilization rates and manure deposition. Thus, mobile housing (when applicable) can also support a uniform distribution of manure. Other considerations include optimizing feed management by adjusting feed quantities according to animal behavior and their seasonal forage intake and changing feed composition by substituting overseas ingredients for locally produced ones.

Several research needs have also emerged, especially in standardizing LCA methods for handling multifunctionality, estimating C-seq, and tracking nutrient pathways to improve cross-study comparability. More field and experimental data are also needed to further substantiate the interactions between animal behavior and tree-dense environments (see e.g., Jakobsen *et al.*, 2019; Manevski *et al.*, 2019). Key questions include whether trees enhance forage intake or feed availability (and to what extent), or if similar outcomes could be achieved in treeless (grass-only) settings. Understanding these dynamics would provide valuable insights into the contributions of silvopastoral configurations.



Chapter 5

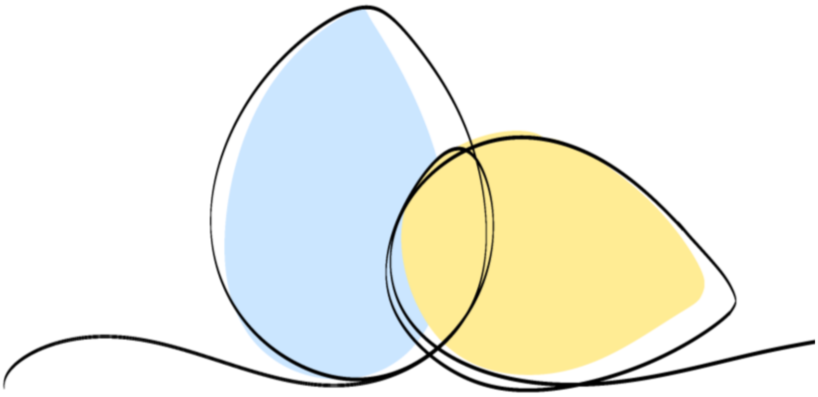
5. Conclusion

This chapter provides conclusions regarding the key findings of the PhD, linking them to the central research objective. The conclusions reflect on the insights obtained from the multiple-case study comparisons on the topic of life cycle assessment and agroforestry, including the broader implications for the global food system.

5.1. Concluding remarks

The central objective of this PhD was to explore the use of the LCA methodology as a tool for quantifying the net environmental and climate impacts associated with food from AFS. The multiple case study comparison shows that, at the farm-gate level, AFS has better environmental performance than non-AFS for certain crops and animal-based products, with eggs also showing favorable outcomes per kg product. However, some AFS show greater environmental impacts per kg product, particularly for some dairy products and apples. The mixed outcomes underscore that the environmental performance of AFS is not consistently better or worse but highly context-dependent, influenced by factors such as the type of crop or animal used, the specific management practices employed, and the methodological frameworks applied to assess them. Thus, generalizations should be approached with caution, and further LCA studies for different agroforestry configurations are essential to increase the number of observations. The findings also show that the implementation of AFS must be carefully managed to enhance opportunities and show that shorter and local value chains can enhance benefits for organic products in AFS.

An important insight from this thesis is the variability introduced by different modeling approaches and methodological choices in LCA, which significantly influences the assessment. Methodological variability in handling multifunctionality and estimating C-seq complicates cross-study comparisons, suggesting that a more standardized approach is needed. Furthermore, most of the potential environmental outcomes often cited in the agroforestry literature are not fully captured from an LCA perspective. The partial inclusion of relevant environmental interactions strongly suggests that a more nuanced evaluation is required to better capture the positive and negative impacts of AFS and their respective configuration across the food value chain. Future research could benefit from experimental data on animal behavior in tree-dense environments, methodological developments in the LCA domain, increased collaboration with agronomists, and a better alignment with planetary health goals.



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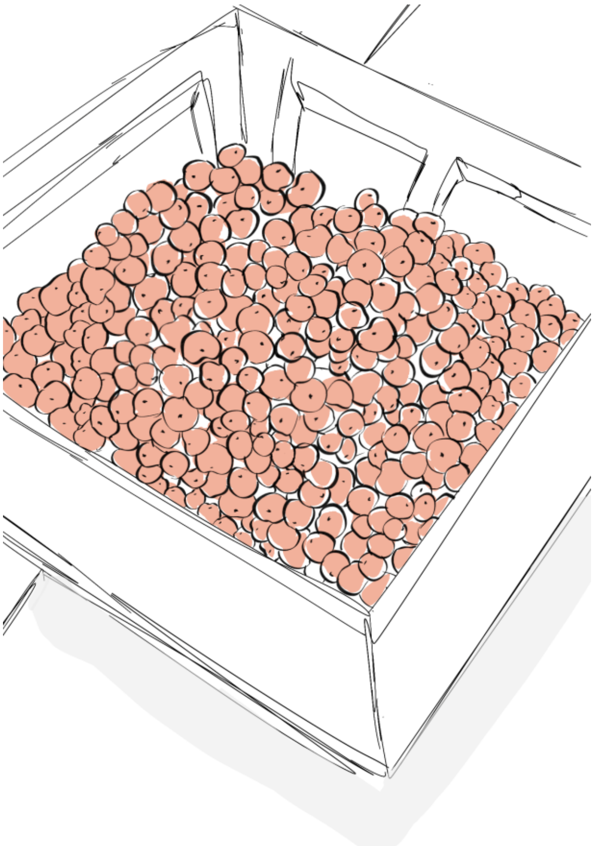

Appendices

Appendix A

A.1. Paper I

How does Life Cycle Assessment capture the environmental impacts of agroforestry? A systematic review

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Review

How does Life Cycle Assessment capture the environmental impacts of agroforestry? A systematic review

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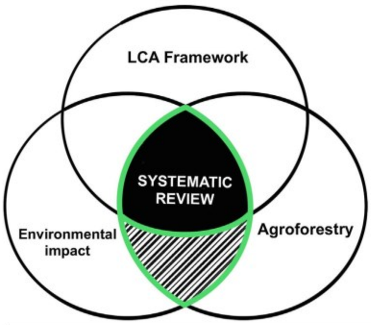
HIGHLIGHTS

- Life Cycle Assessments (LCA) of agroforestry systems (AFS) were systematically reviewed.
- LCAs in temperate regions are mainly located in southern Europe.
- The LCA literature on AFS is scarce and climate change is the most studied impact.
- Other environmental outcomes from the AFS literature are often not captured in LCAs.
- Methodological challenges include multifunctionality and carbon sequestration, among others.

GRAPHICAL ABSTRACT



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ABSTRACT

In this paper, a systematic review approach was used to evaluate how environmental Life Cycle Assessment (LCA) has been applied in agroforestry in the context of food systems. This review was used as the basis for discussing methodological issues in the LCA framework for agroforestry systems (AFS) and relevant environmental outcomes in the agroforestry literature. A total of 32 LCAs in 17 countries identified in four databases and spanning a decade form the basis for this paper. Studies were selected based on pre-defined inclusion criteria and followed established guidelines and a review protocol. Qualitative data were extracted and categorized into multiple themes. Results were quantitatively synthesized for the four phases of the LCA for each individual agroforestry practice (i.e., based on its structural composition). Results showed that around half of the selected studies are located in tropical climates, the rest being in temperate climates, predominantly in Southern Europe. Studies primarily used a mass functional unit and rarely included post-farm gate system boundaries. Almost half of the studies account for multifunctionality, and most allocation methods were based on physical properties. Climate change had the greatest coverage from all impact categories with some variations within milk, meat, and crop production systems. Methodological issues were related to limited system boundaries, few impact categories, and differing functional units and multifunctionality approaches. The identified effects of AFS on biodiversity, climate change mitigation, water, soil, pollination, and pest and disease were only partially documented or not analyzed in the LCA studies or the LCA framework. Gaps in knowledge and limitations of the present review were discussed. Further methodological improvements remain necessary to determine the net environmental effects of food products resulting from individual AFS, especially within the area of multifunctionality, carbon sequestration, and biodiversity.

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Contents

1. Introduction

Proper management of food systems is crucial for preserving healthy ecosystems and providing essential nutrients for human health and well-being (Neven, 2014). However, today's dominant production model is negatively affecting the environment through deforestation, greenhouse gas (GHG) emissions, and biodiversity loss, among others (Horrigan et al., 2002; Notarnicola et al., 2017). To address the environmental challenges posed by the status-quo, different archetypes for environmentally responsible and eco-friendly practices have been proposed (Brinken et al., 2022; Garnett, 2014), which have given rise to various alternative production models (see e.g., Garcia-Oliveira et al., 2021).

Agroforestry, a historical farming practice, has long been promoted as one alternative model (Jose, 2009; Wilson and Lovell, 2016). Agroforestry is a collective term that describes the integration of woody perennials with livestock and crop production (FAO, 2013). According to Nair (1993), agroforestry can be structurally classified into three practices encompassing (i) woody perennials and arable crop production (agrosilvicultural), (ii) woody perennials and livestock production (silvopastoral), and (iii) woody perennials, livestock, and arable crop production (agrosilvopastoral). Unlike specialized systems, agroforestry typically has no clear segregation between arable crops, woody perennials, or pasture lands.

The environmental impacts of agroforestry have been documented in the scientific literature with convergent and divergent results. For example, while agroforestry has the potential to mitigate climate change by increasing carbon sequestration (particularly when transforming degraded land into improved fallow in tropical climates, as shown in Feliciano et al., 2018), the decomposition of organic matter from thinning and pruning can challenge long-term sequestration by releasing carbon (Lorenz and Lal, 2014). Similarly, although agroforestry (e.g., riparian buffers) can improve water quality compared to intensive agriculture (Jose, 2009; Kremer, 2021), it can potentially deplete freshwater sources in water-scarce areas due to the introduction of non-native species, as observed in the Cerrado biome (Ollinaho and Kröger, 2021). Concurrently, tree-animal-crop interactions can be influenced by natural (e.g., resource competition) and anthropogenic (e.g., fertilization) activities to or from the environment (Zhang et al., 2007). These interactions will likely change over time due to their dynamic nature at different spatial and temporal scales. For example, some interactions may be complementary or positive in the early stages but become competitive or negative at a later stage (or vice-versa) (Rao et al., 1997; Rudd et al., 2021; Smith et al., 2013). Furthermore, typical levels of analysis in agroforestry research are on the plot or field level and often in one study site only (Fagerholm et al., 2016), which overlooks the broader picture within its current scope. Thus, conclusions drawn at the plot or field level may contradict those drawn on the broader system, given that upstream impacts can be far greater than those occurring in the agricultural field (Hellweg and Canals, 2014), highlighting critical trade-offs important to, for example, the land-sparing or sharing discussion (Collas et al., 2022).

The question of whether and to what extent agroforestry benefits the environment could be answered through a Life Cycle Assessment (LCA). LCA is a standardized methodology for quantifying the environmental impacts of a product or service throughout its entire life cycle (Baumann and Tillman, 2004; ISO, 2006a, 2006b). Through the LCA framework, the focus of attention is extended to the systems level, i.e., the agroforestry food value chain, hereafter referred to as agroforestry systems (AFS) (Onat et al., 2017). However, the standard methodological approaches in LCA for analyzing single-species systems (e.g., monocultures or specialized husbandry) face difficulties in capturing the more complex processes involved in integrated systems, such as agroforestry (Haas et al., 2000). It is therefore important to evaluate what are the methodological choices in the LCA literature of AFS and whether the existing choices under the LCA framework capture key environmental outcomes grounded on the general agroforestry literature. Environmental outcomes are defined in this paper as the biophysical impacts on the natural environment.

To the best of found knowledge, a comprehensive and quantitative overview of existing peer-reviewed LCA studies on AFS is lacking. Thus, the objective of this paper is threefold. First, to systematically identify, select and review a collection of LCA studies on agroforestry in the context of food systems. Second, to synthesize the results and methodological choices across the four LCA phases. Third, this review is used as the basis for discussing results against the most recent findings in Köthke et al. (2022) on the environmental outcomes of agroforestry and methodological issues within the LCA framework. As such, the primary and secondary research questions are: (1) What evidence exists in process-based LCAs on the methodological choices and environmental impacts attributed to global food products from AFS? and (2) What are the key methodological issues of capturing relevant environmental outcomes of AFS in process-based LCAs compared to the existing literature on environmental impacts from agroforestry?

2. Methods

2.1. Literature review

2.1.1. Identification of studies

The literature review was based on pre-defined inclusion criteria and followed established guidelines and a review protocol provided by Zumsteg et al. (2012) and Bilotta et al. (2014) (Supplementary Table S1). A total of 350 studies were identified, and 249 were screened for relevance at the topic level (i.e., title, abstract, and keywords). Studies were extracted from online portals and article databases relevant to this paper's subject, namely, *Web of Science, Scopus, AGRIS,* and *CAB Direct*. Subsequently, a manual search was conducted based on relevant references cited in the selected studies (Fig. 1).

After a preliminary review scoping (Supplementary Table S2), two search strings and logical operators (in databases that allowed it) were applied in order to identify studies (Table 1). The first string refers to the terms describing LCA methodology and related methods building on existing ISO standards for LCA (e.g., carbon footprint). The second string refers to the definitions linked to the structural composition of AFS based on Nair (1993), Atangana et al. (2014a), Torralba et al. (2016) and Eksvärd (2016) descriptions. Since the search aimed to find all available studies, the geographical scope was global.

2.1.2. Selection criteria

Studies were selected under the guidelines of Zumsteg et al. (2012) based on the criteria shown in Table 2 and explained in Supplementary Table S3. All life cycle impact categories were of interest to this paper.

2.2. Data analysis

Qualitative and descriptive data from all eligible studies were extracted and categorized into multiple themes using Covidence Systematic Review

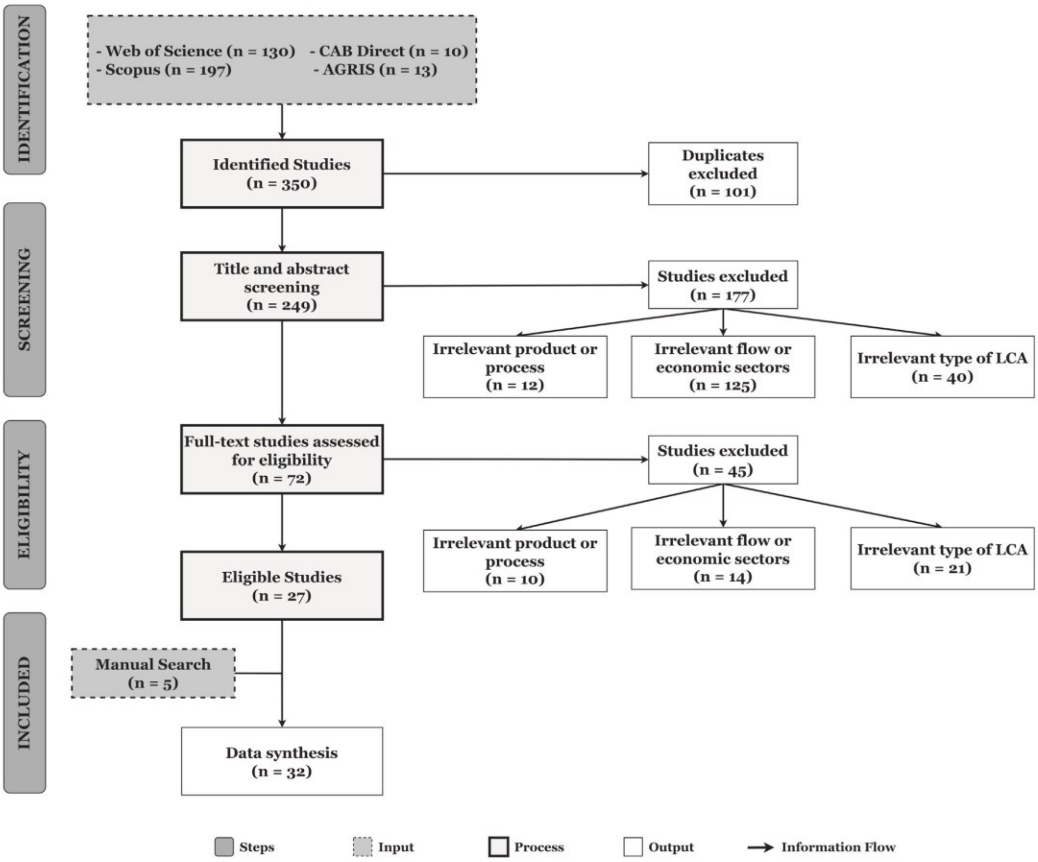

Fig. 1. Systematic review process.

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Table 1
Search terms applied for relevance at the topic level. String 1 refers to the methodology, and string 2 to the agroforestry definitions.

Search string	Search terms
1	"Life Cycle Assessment" OR "Life Cycle Analysis" OR "life cycle approach" OR "life cycle perspective*" OR "life cycle method*" OR "carbon footprint" OR "C footprint" OR "environmental footprint"
2	Agr*forest* OR silvopast* OR silvoarable OR agr*silvopast* OR agr*silvicultur* OR aquaforest* OR "silvofisher*" OR "apiculture with trees" OR entomoforest* OR dehesa OR montado OR meriagos OR taungya OR pannage OR streuobst OR "pré-verger" OR Joualle OR Hauberg OR pomarada* OR "improved fallow*" OR "tree fallow*" OR "alley cropping" OR "multilayer tree*" OR "multipurpose tree*" OR "home garden*" OR shelterbelt* OR windbreak* OR hedgerow* OR "tree hedge*" OR "live hedge*" OR "liv* fence*" OR "protein bank*" OR parklands OR hedge* OR "shad* crop*" OR "riparian woodland*" OR "buffer strip*" OR "riparian buffer" OR "wood* pasture" OR "forest farm*" OR "farm woodland*" OR "farm forest*" OR "farm tree*" OR "shelter belt*" OR "tree system*" OR "forest grazing" OR "grazed forest*" OR hillside OR "linear strip*" OR "mixed forest*" OR "tree garden*" OR "mixed wood" OR "tree intercrop*" OR "tree based" OR "woodland chicken*" OR "food forest*" OR "fodder tree*" OR "tree outside forest*" OR (crop* AND (tree* OR forest*) AND (livestock OR animal)

Software (Veritas Health Innovation, Australia). Covidence is a web-based tool that helps streamline literature reviews. Guidelines on data coding strategies by Yin (2009) were followed. Second- or third-order themes were grouped into a single first-order theme and merged into aggregated categories to create a condensed table (Supplementary Table S4). The core coding principle involved examining important content – paragraphs and sentences (third-order themes) – and classifying it with a word that summarizes its content (second- and first-order themes), reducing large amounts of information in categories and making it conceptually accessible for data extraction. The following aggregated categories were developed: (1) agroforestry practices, (2) agroforestry components, (3) life cycle impact assessment (LCIA) at the midpoint and endpoint level and 'other', and (4) life cycle impact contributions. The extracted data was then exported to Excel (version 2202). A standardized spreadsheet was used to organize further and synthesize details about each selected study.

Only agroforestry-related data were extracted from each study. Therefore, technical and qualitative data on non-agroforestry activities unlinked to AFS, such as monocultures, specialized farms, grazing systems, or rangelands, were not accounted for in the synthesis of this paper (except for basic information, see Supplementary Table S5). Descriptive data were based on basic information and the LCA framework, namely: (1) Goal and scope definition, (2) life cycle inventory, (3) LCIA and (4) interpretation.

3. Results

Only 8 % of the 350 identified studies (without manual search) are relevant to this paper. Of the excluded studies (92 %), 29 % are duplicates and 6 %, 40 %, and 17 % are studies with irrelevant products, economic sectors, or types of LCA, respectively. Studies ($n = 350$) are primarily identified through Scopus (56 %) and Web of Science (37 %). Selected studies ($n = 27$) are predominantly published in the Journal of Cleaner Production (37 %) and the Science of the Total Environment (15 %). Five additional manuscripts have been identified through a manual search. Thus, a total of 32 peer-reviewed studies are synthesized in this paper (Table 3). An extended table is shown in Supplementary Table S6.

3.1. Phase 1: Goal and scope

The Functional Unit (FU) differs between studies, and some use multiple ones. The most frequent FU in all AFS is based on mass, followed by area, economic, composite, and energy features (Fig. 2a). The system boundary in all studies is mainly from cradle-to-farm gate, following a cradle-to-retail or cradle-to-grave approach (88 %, 9 %, and 3 %, respectively) (Fig. 2b). The selected studies analyze a range of AFS, and two examine several interventions simultaneously. In total, agrosilvicultural, agrosilvopastoral, and silvopastoral systems are included in 13, 11, and 10 studies, respectively. Following the global trend in the general agroforestry literature (Castle et al., 2022), the highest density of selected studies in the last decade can be found for agrosilvicultural systems, followed by agrosilvopastoral and silvopastoral (Fig. 2c). The majority of the selected studies (25 %) are from 2022, followed by 2020 (22 %) and 2021 (16 %) (according to their year of publication).

Different and multiple agroforestry components have been assessed simultaneously in the selected studies (Fig. 2d). Regarding the tree component, bananas, olives, and plantains are the species most representative of the fruit trees category. Non-timber trees include fiber, tree nuts, and to a lesser extent, latex products obtained from cork oak, holm oak, and rubber trees, respectively. 'Other' trees are characterized by shrubs, forest meadows, forest pastures, medicinal trees, or native forests. Many studies do not specify the type of shade trees, but some include Inga and leguminous species. Eucalyptus and willow trees are the most popular timber and energy trees, respectively. Regarding the crop component, barley, wheat, corn, and vetch are the most common food/feed crops. To a lesser extent, beans, pea, and sorghum are also mentioned, among others. Common cash crops are cocoa and coffee. In terms of animals, cattle, sheep, and goats are most common in the ruminant category, followed by pigs and chickens in the monogastric and poultry categories.

According to Kottek et al. (2006) climate classification, studies are almost evenly distributed between tropical (16 studies) and temperate (18 studies) zones and just one is located in an arid zone (Fig. 2e). The overall literature is predominantly European-centric, with Southern Europe (i.e., Spain and Italy) more represented. The most dominant studies in the Americas are from Brazil and Ecuador. A handful of studies are located in West Africa and the Southern and Southeast Asian regions (i.e., Ghana, India, and Indonesia).

3.2. Phase 2: life cycle inventory

Among the selected studies ($n = 32$), only 15 address multifunctionality explicitly through different approaches (Table 4). Three studies examining multifunctionality use more than one approach, whereas 12 use only one. Studies addressing multifunctionality are primarily on silvopastoral systems. Multifunctionality is mostly handled by allocation methods based on physical properties ($n = 11$) or biophysical causality

Table 2
Inclusion and exclusion criteria. (1) relevant product or process, (2) relevant flow or economic sectors, (3) relevant type of LCA, and (4) relevant type of article.

ID	Criteria	Inclusion	Exclusion
1	Relevant product or process	Only studies with a focus on food provision were chosen	Studies with primary activities other than food provisioning (e.g., biomass-to-bioenergy systems) were excluded
2	Relevant flow or economic sectors	Only studies focused on agroforestry activities were selected	Studies focused exclusively on forestry, conventional agriculture, or other economic sectors (e.g., energy) were discarded
3	Relevant type of LCA	Only processed-based LCAs or studies that explicitly mention LCA in their methodology were included	Stand-alone input-output models, footprints (e.g., Cool Farm Tool, TropiC Farm Tool), economic or social assessments or other non-LCA methodologies were omitted
4	Relevant type of article	Only full-text primary and published journal articles in English were considered (the search was not limited to peer-reviewed articles only)	Non-English and non-journal articles (e.g., conference papers, reports) or review articles were excluded

Table 3
Summary of selected LCA studies.

ID	Source	Location	Type of AFS	Functional unit	System boundary	Impact category[a]
1	Acosta-Alba et al. (2020)	Colombia	Agrosilvicultural Agrosilvopastoral	Mass Area Economic	Cradle-to-gate	CC, EU, AC, RD, and EC
2	Armengot et al. (2021)	Bolivia	Agrosilvicultural	Mass	Cradle-to-gate	CC, EU, AC, RD, EC, and POC, PO or POF
3	Bianchi et al. (2021)	Indonesia Ghana Ecuador	Agrosilvicultural	Mass Energy	Cradle-to-grave	CC, EU, AC, RD, CED, and POC, PO or POF
4	Brook et al. (2022)	Costa Rica	Silvopastoral	Mass	Cradle-to-gate	CC
5	Caicedo-Vargas et al. (2022)	Ecuador	Agrosilvopastoral	Mass Area	Cradle-to-gate	CC, EU, AC, RD, CED, EC, HT, and POC, PO or POF
6	Caputo et al. (2020)	Italy	Agrosilvicultural	Mass Area	Cradle-to-grave	CC and RC
7	Costa et al. (2018)	Brazil	Agrosilvopastoral	Composite	Cradle-to-gate	CC, EU, AC, RD, EC, LU, RC, and POC, PO or POF
8	Crous-Duran et al. (2019)	Portugal	Agrosilvicultural	Mass	Cradle-to-gate	CC
9	de Figueiredo et al. (2017)	Brazil	Agrosilvopastoral	Mass	Cradle-to-gate	CC
10	Doddabasawa et al. (2020)	India	Agrosilvicultural	Area	Cradle-to-gate	CC
11	Duffy et al. (2021)	Costa Rica	Silvopastoral	Mass	Cradle-to-gate	CC
12	Eldesouky et al. (2018)	Spain	Agrosilvopastoral	Mass	Cradle-to-gate	CC
13	Escribano et al. (2022)	Spain	Agrosilvopastoral	Mass Area	Cradle-to-gate	CC
14	Gutiérrez-Peña et al. (2019)	Spain	Agrosilvopastoral	Mass	Cradle-to-gate	CC
15	Horrillo et al. (2021)	Spain	Agrosilvopastoral	Area	Cradle-to-gate	CC
16	Horrillo et al. (2020)	Spain	Agrosilvopastoral	Mass Area	Cradle-to-gate	CC
17	Lamnatou et al. (2022)	Spain	Agrosilvopastoral	Mass	Cradle-to-gate	CC, EU, RD, AC, HT, EC, CED, LU HH, E, R, POC, PO, or POF
18	Lehmann et al. (2020)	Brazil Italy	Silvopastoral Agrosilvicultural	Mass Area Economic	Cradle-to-gate	CC, EU, AC
19	Livingstone et al. (2021)	Ireland	Silvopastoral	Area	Cradle-to-gate	CC, EU, GEP
20	Martinelli et al. (2019)	Brazil	Agrosilvicultural	Area	Cradle-to-gate	CC
21	Mazzetto et al. (2023)	New Zealand	Silvopastoral	Mass	Cradle-to-grave	CC
22	Paolotti et al. (2016)	Italy	Silvopastoral	Mass	Cradle-to-gate	HH, E, R
23	Parra-Paitan and Verburg (2022)	Ghana	Agrosilvicultural	Mass	Cradle-to-gate	CC, EU, EC, AC, HT, E, HH
24	Pérez-Neira et al. (2020)	Ecuador	Agrosilvicultural	Mass	Cradle-to-retail	CC, EU, RD, AC, HT, EC, LU, CED, and POC, PO or POF
25	Raschio et al. (2018)	Peru	Agrosilvicultural	Mass Area	Cradle-to-gate	CC
26	Reyes-Palomo et al. (2022)	Spain	Silvopastoral	Mass	Cradle-to-gate	CC
27	Ripoll-Bosch et al. (2013)	Spain	Agrosilvopastoral	Mass	Cradle-to-gate	CC
28	Rocchi et al. (2019)	Italy	Silvopastoral	Mass	Cradle-to-gate	HH, E, R
29	Rowntree et al. (2020)	USA	Silvopastoral	Mass	Cradle-to-gate	CC
30	Ruiz-Llontop et al. (2022)	Peru	Silvopastoral	Mass	Cradle-to-gate	CC
31	Tziolas et al. (2022)	Greece	Agrosilvicultural	Area	Cradle-to-gate	CC
32	Utomo et al. (2016)	Indonesia	Agrosilvicultural	Mass	Cradle-to-gate	CC, AC, EU

[a] CC = Climate Change; AC = Acidification; EU = Eutrophication; RD = Resource Depletion; RC = Resource Consumption; HT = Human Toxicity; EC = Ecotoxicity; CED = Cumulative Energy Demand; GEP = Gross Energy Production; LU = Land Use; POC, PO, or POF = Photochemical Oxidant Creation, Photochemical Oxidation or Photochemical Ozone Formation; HH = Human Health; E = Ecosystems; R = Resources.

(n = 2). Some studies do not apply allocation methods but instead address multifunctionality through subdivision (n = 2) or other approaches (n = 4), such as allocation of emissions through spatial distribution and multifunctional units (i.e., composite or multiple FUs). In livestock related AFS (silvopastoral and agrosilvopastoral), economic allocation based on physical properties is mostly used.

3.3. Phase 3: life cycle impact assessment

In total, 18 studies focus only on climate change impacts, while 14 analyze more than one impact category. Two studies include both midpoint and endpoint methods. Thus, 30 and 4 studies conduct their LCIA at the midpoint and endpoint levels, respectively. In Table 5, the dark color coding indicates that the number of studies examining a particular impact category is higher than the average of the total number of studies conducted across all agroforestry groups. At the midpoint level (n = 30), Climate Change, Eutrophication, and Acidification were the three main impact categories chosen by 100 %, 37 %, and 33 % of the mentioned studies, respectively. Other reported impact categories were ecotoxicity, resource depletion, and Photochemical Oxidant Creation (POC), Photochemical Oxidation (PO) or Photochemical Ozone Formation (POF). To a lesser extent, Cumulative Energy Demand (CED), human toxicity, land use, and resource consumption were included, among others. At the endpoint level (n = 4), indicators related to the areas of protection, such as human health, ecosystems, and resources, are given almost equal consideration.

Regarding the environmental impact of AFS, comparing the LCA results is challenging due to the diversity of methods (see more in Section 4.2). The objective is, therefore, to conduct a quantitative analysis to have an overview of the reported LCA results and show the variations among production systems, including main results and scenario analysis (excluding values reported in sensitivity analysis). Supplementary Table S7 provides a complete listing of values for each study. Only midpoint results on climate change are included in this sub-section since 94 % of the total studies account for them (Table 5). Considering that 26 and 28 of the total studies use a mass FU and cradle-to-farm gate system boundary, respectively, values have been selected based on these criteria (Fig. 2a and Fig. 2b). Hence, a total of 82 values (silvopastoral, n = 36 values; agrosilvopastoral, n = 38 values; agrosilvicultural, n = 8 values) for climate change have been extracted from 18 studies and converted to a common unit (i.e., kg CO_2eq). Depending on the type of AFS, the FU can refer to different mass units and crop or animal species. Thus, this paper categorizes values in three contrasting production systems based on the 18 studies. Namely, a milk production system using the unit of kg CO_2eq kg^{-1} Fat Protein Corrected Milk (FPCM) of cow and goat (n = 5 studies) (Fig. 3a), a crop production system in kg CO_2eq kg^{-1} product of cocoa, coffee, and olive (n = 6 studies) (Fig. 3b), and a meat production system in kg CO_2eq kg^{-1} live weight (LW) of beef, calf,

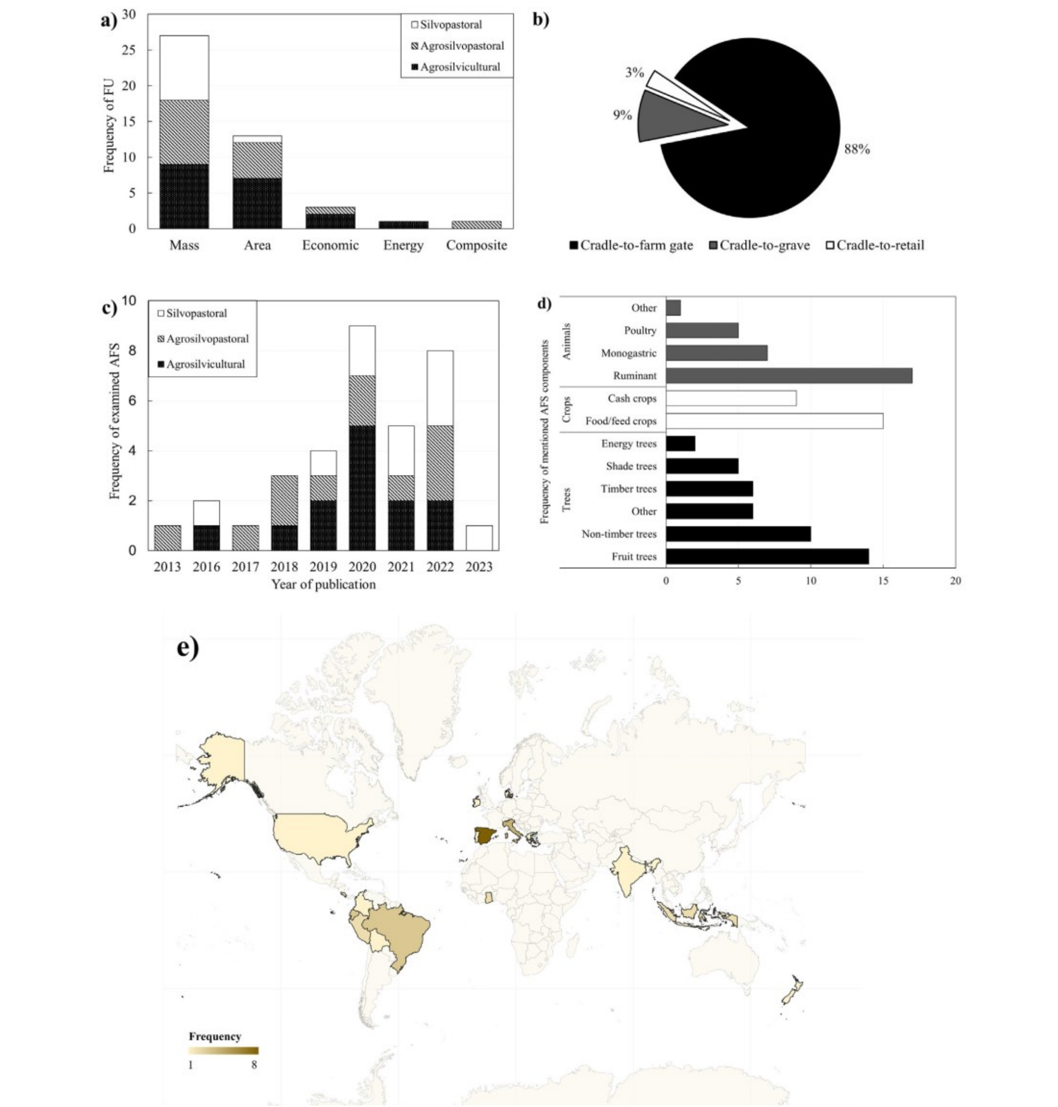

Fig. 2. a) Frequency of the Functional Unit (FU) in the selected studies (n = 32); b) System boundary in the selected studies (n = 32); c) Frequency of AFS examined from 2013 and 2023 (year of publication) (n = 32); d) Frequency of AFS components in the selected studies (n = 32); e) Geographical location where LCA studies are conducted (n = 32).

sheep (including lamb), and pig (n = 8 studies) (Fig. 3c). It should be noted that one study reports values for two production systems (Horrillo et al., 2020) and that studies reporting only carcass weight have not been included (Rowntree et al., 2020). Also, values from Mazzetto et al. (2023) are included in the analysis, as they also report data from cradle-to-farm gate, as specified in Table S6. Only the maximum and minimum values from studies reporting their results within an inclusive range have been included in Fig. 3 (Brook et al., 2022; Gutiérrez-Peña et al., 2019; Lamnatou et al., 2022; Reyes-Palomo et al., 2022). Two studies have been excluded due to the extreme values (Crous-Duran et al., 2019; Raschio et al., 2018).

Fig. 3 shows heterogeneity within the results of each production system, with the highest variability in meat systems, as observed in the negative values among silvopastoral and agrosilvopastoral groups (Fig. 3c), especially for beef and calf. The minimum and maximum rounded values for meat production systems range between −28 to 18, −22 to 31, 1 to 26, and − 3 to 6 kg CO_2eq kg^{-1} live weight (LW) of beef, calf, sheep (including

Table 4
Number of studies reporting on multifunctionality approaches and the applied allocation method.

Methods	Agrosilvicultural (n = 4)	Agrosilvopastoral (n = 5)	Silvopastoral (n = 6)
Allocation (biophysical causality)			
Mass	0	1	1
Allocation (physical properties)			
Economic	1	2	3
Energy content	1	0	2
Mass	1	0	1
No allocation			
Subdivision	0	0	2
Other			
Multifunctional unit	0	2	0
Spatial distribution	2	0	0

lamb) and pig, respectively. Regarding milk production systems, rounded values are between 0.5 to 3 and 0 to 3 kg CO_2eq kg^{-1} FPCM of cow and goat milk, respectively. In crop production systems, minimum and maximum rounded values are between −0.03 to 4, 3 to 5, and 0.1 to 0.6 kg CO_2eq kg^{-1} product of cocoa, coffee, and olive, respectively. Variations among and within agroforestry groups and production systems may be due to the asymmetric and uneven value distribution found in the 18 studies (e.g., only 8 values pertain to agrosilviculture), as well as the fact that values from three studies include standard deviations (i.e., Brook et al., 2022; Gutiérrez-Peña et al., 2019; Reyes-Palomo et al., 2022).

3.4. Phase 4: interpretation

As part of the life cycle interpretation phase, sensitivity and uncertainty analysis are recommended by ISO (2006a). From all studies, 19 % and 3 % perform sensitivity and uncertainty analyses, respectively. An important aspect of sensitivity analysis is determining which input variables may influence the output variance (Groen, 2019). Costa et al. (2018) conducted a sensitivity analysis to identify the influence of zinc in cattle feed and fertilizers on sustainability scores. In Paolotti et al. (2016), benefits typically excluded from LCAs were tested through a sensitivity analysis, such as feed ingested through grazing in tree-based pastures. Although allocation choices can strongly affect the LCA results (Groen, 2019), only Bianchi et al. (2021) and Mazzetto et al. (2023) performed a sensitivity analysis on different allocation approaches (i.e., mass and energy). Furthermore, none of the studies conducting sensitivity analyses analyzed uncertainty. Although uncertainty analysis is important for interpreting the LCIA phase, only Brook et al. (2022) estimated the uncertainty of emission factors and carbon sequestration using error propagation. While country-specific data is recommended by international guidelines (e.g., specific manure management or nitrogen excretion rates), many of the selected studies use default equations and ranges for emission factors. Only Reyes-Palomo et al. (2022) and Mazzetto et al. (2023) tested the sensitivity of their final carbon footprint using different values for emissions factors for methane and nitrous oxide, among others.

Although 84 % of studies conducted a contribution analysis, a general pattern cannot be identified due to the wide variety of applied methods, case studies, and AFS. However, there are some repeated processes (e.g., enteric fermentation) that are responsible for the greatest share of climate change, the most reported impact category in all studies (Fig. 4.). It should be noted that when the selected studies reported their contribution analysis for multiple case studies and did not present results for the total sample, only one case study was chosen, albeit the variations were negligible, and thus, representative to the total sample. Regarding agrosilvopastoral and silvopastoral systems, primary hotspots are from direct/indirect field emissions. More specifically, the main sources of emissions are from livestock enteric fermentation and inputs application such as mineral fertilizer. Secondary hotspots are from raw material extraction, such as off-farm feed production, and direct/indirect field emissions, such as nitrous oxide emissions from manure and soil management. Machinery operation, farm emissions (not specified), and agrochemical and seed production are equally important primary hotspots for agrosilvicultural systems. Transportation, operational, and manufacturing activities are other important but less cited contributors in all systems. The frequency of environmental hotspots identified in selected studies is shown in Fig. 4.

4. Discussion

This section is structured in three parts. First, it summarizes the methodological issues identified in the selected studies and the LCA framework. Second, it builds on the general agroforestry literature where mainstream environmental outcomes of AFS (beyond the LCA domain) are discussed per type (Table 6) according to the review by Köthke et al. (2022) and

Table 5
Frequency of midpoint and endpoint impact indicators analyzed in the selected studies, per AFS and in total. Dark color code: the number of studies considering this impact category is above the average of the total number of studies for all agroforestry groups.

Categories[a]	Agrosilvicultural (n = 13)	Agrosilvopastoral (n = 11)	Silvopastoral (n = 8)	Total
Midpoint				
Acidification	6	4	1	10
CED	2	2	0	4
Climate Change	13	11	8	30
Ecotoxicity	4	4	0	7
Eutrophication	7	4	2	11
Human toxicity	2	2	0	4
Land use	1	2	0	3
POC, PO or POF	3	3	0	6
Resource consumption	1	0	0	1
Resource depletion	4	4	0	7
Endpoint				
	Agrosilvicultural (n = 1)	Agrosilvopastoral (n = 1)	Silvopastoral (n = 2)	Total
Ecosystems	1	1	2	4
Human health	1	1	2	4
Resources	0	1	2	3

[a] Four 'Other LCIA' indicators (described in supplementary Table S4) used in four studies (Armengot et al., 2021; Costa et al., 2018; Livingstone et al., 2021; Rocchi et al., 2019) were not included in this table since neither a midpoint nor an endpoint method was applied.

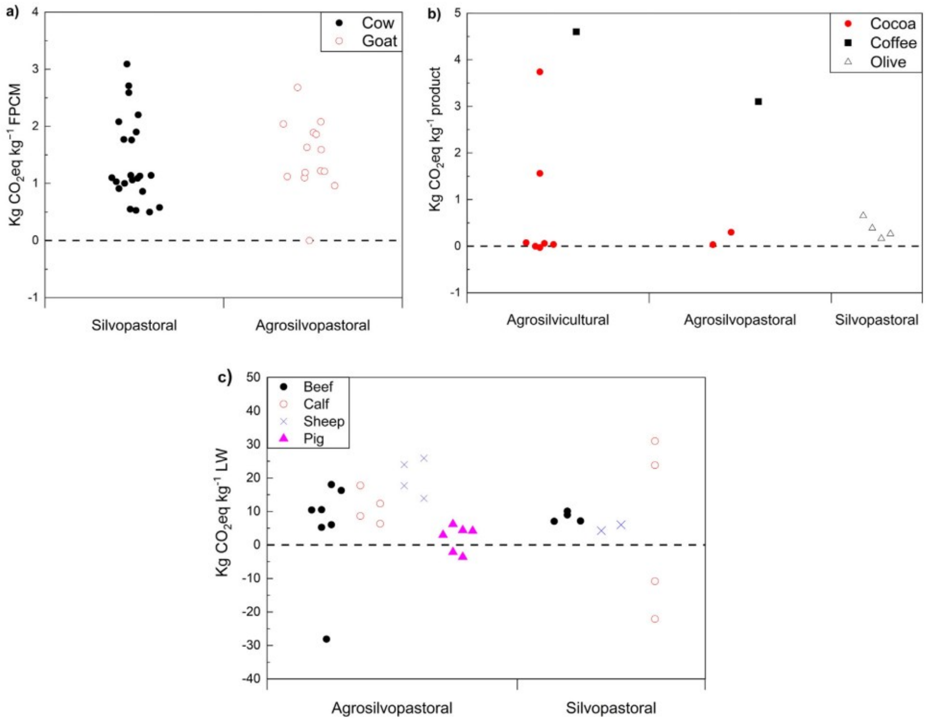

Fig. 3. Scatter plots by groups (i.e., agroforestry type) and its impacts on climate change at the midpoint level categorized in three contrasting production systems based on 82 values from 18 studies with cradle-to-farm gate system boundaries and mass functional units: a) milk production systems in kg CO_2eq kg^{-1} Fat Protein Corrected Milk (FPCM) color coded for cow (black) and goat (red); b) crop production system in kg CO_2eq kg^{-1} product color coded for cocoa (red), coffee (black), and olive (white); c) meat production system in kg CO_2eq kg^{-1} live weight (LW) color coded for beef (black), calf (red), sheep (including lamb) (blue) and pig (pink).

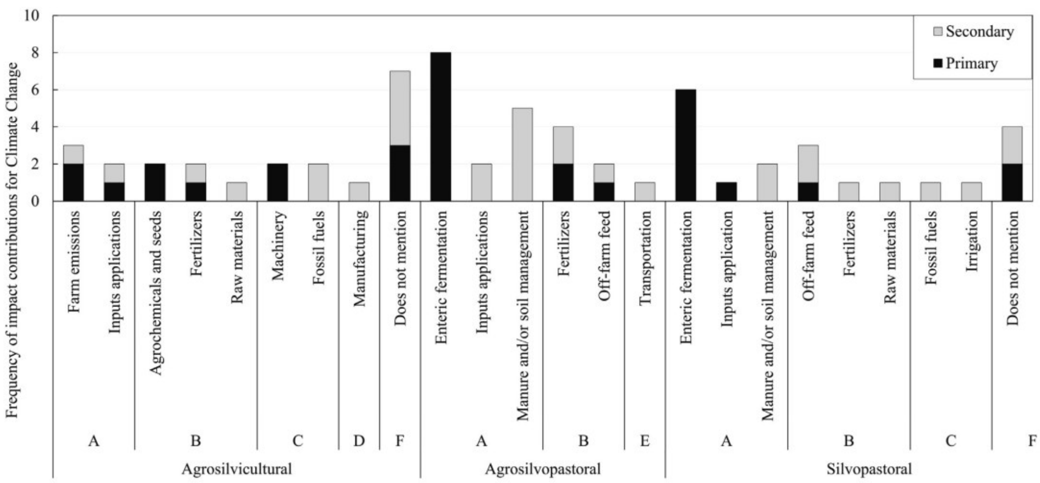

Fig. 4. Frequency of environmental hotspots for the climate change impact category (n = 27) where: A = Direct/indirect field emissions; B = Raw material extraction; C = Operation; D = Manufacturing; E = Transportation; F = Does not mention.

Table 6
Non-exhaustive overview of environmental outcomes linked to agroforestry. Modified from Köthke et al. (2022). Indicators reported in the review by Köthke et al. (2022) have been used as the description.

Environmental outcome	Description
Biodiversity	Species richness, abundance, composition, or functional or taxonomic diversity
Climate change mitigation	Above- and below-ground carbon sequestration
Water	Regulating services and retention (soil hydrological properties, such as field capacity, infiltration rate, soil moisture, surface runoff, soil water content or porosity, and runoff of sediment)
	Quality (nutrients or fecal bacteria and water purification)
Soil	Chemical properties (soil fertility, nitrogen, phosphorus, pH, and nutrient cycling)
	Biological properties (soil macro, meso, or microfauna)
	Physical properties (erosion and structural indicators, such as bulk density, aggregate stability, or macroaggregates)
Pest and disease control	Abundance, density or diversity of natural enemies/predators
Pollination	Pollinator richness and abundance

against the LCA framework. Finally, it examines the limitations of the present review.

4.1. Geographical representativity, functional unit (FU) and multifunctionality

Geographically, most studies are conducted in Southern Europe or South America, suggesting a strong need for regionalized and local datasets to understand better agroforestry activities across different climate zones (Finkbeiner et al., 2014; Udo De Haes, 2004) since macroclimate conditions and geography can affect AFS. For example, the effect can be different in areas with cooler temperatures and higher precipitation, such as the Alpine, Boreal, Atlantic, and Continental regions (Torralba et al., 2016).

The choice of the FU relies on the LCA's goal and scope (ISO, 2006b). However, it is difficult to choose a FU when AFS have multiple farming goals, such as producing fiber, energy, or feed and food crops. In addition, as Haas et al. (2000) noted, a FU based on mass may be relevant for global impacts such as climate change or specific sustainability archetypal goals such as eco-efficiency (Bjørn and Hauschild, 2013; Brinken et al., 2022; Garnett, 2014). However, local and regional environmental impacts may be more important to area-related FU since agricultural impacts (e.g., nitrogen leaching) must be minimized regardless of production efficiency (Haas et al., 2000). Furthermore, some argue that nutrient supply is an essential component in food systems that should be considered in the FU (e.g., nutrient density) (Weidema and Stylianou, 2020).

Moreover, multifunctionality in AFS is partially handled or not included in the selected studies. There are many products and services that can result from AFS, which are methodologically difficult to handle. Multifunctionality approaches can differ according to the life cycle stage (Mazzetto et al., 2023). For example, a mass or economic allocation approach was commonly used in the processing phase (Bianchi et al., 2021; Mazzetto et al., 2023), while an economic allocation approach was most common in the production phase (Gutiérrez-Peña et al., 2019; Mazzetto et al., 2023; Reyes-Palomo et al., 2022; Ripoll-Bosch et al., 2013; Ruiz-Llontop et al., 2022; Utomo et al., 2016). In addition to the multiple products delivered by AFS, some ecosystem functions may be undervalued or not valued at all in the LCA framework. Only one study accounted for the multifunctionality of ecosystem services in their analysis (Ripoll-Bosch et al., 2013).

4.2. Impact categories and inventory data

The direction and magnitude of the results of each impact category will vary among the selected studies (as shown in Fig. 3) depending on the assumptions, such as emission factors for organic or conventional feed inputs, water sources for spraying, and outdoor feed intake (see e.g., Paolotti et al., 2016; Costa et al., 2018; Reyes-Palomo et al., 2022), carbon sequestration approaches and time perspective (see e.g., de Figueiredo et al., 2017; Brook et al., 2022), allocation approaches (see e.g., Bianchi et al., 2021; Mazzetto et al., 2023), milk correction factor (see e.g., Gutiérrez-Peña et al., 2019), inclusion of climate-carbon cycle feedback (see e.g., Mazzetto et al., 2023; Reyes-Palomo et al., 2022), type of forcing centered metric (see e.g., Mazzetto et al., 2023), weighting process (see e.g., Rocchi et al., 2019), management practices (e.g., on-farm or off-farm resources), farm design (e.g., selection of animal, crop, or tree species), reference system, such as degraded pasture, non-agroforestry, organic agroforestry, or conventional agroforestry (see e.g., Utomo et al., 2016; de Figueiredo et al., 2017; Reyes-Palomo et al., 2022), and system boundary (for instance, negative values observed in Fig. 3 may be explained because only the emissions from the animal fattening period were considered) (see e.g., de Figueiredo et al., 2017).

Based on the review by Köthke et al. (2022), AFS can potentially contribute to several environmental outcomes, such as those categorized in Table 6. Although agroforestry was assumed to have positive effects on several environmental outcomes in many of the selected studies, 60 % of the studies only assessed the climate change impact category. As such, some studies do not always support their claims with data on specific environmental outcomes in the context in which the LCA is conducted. While there is a legitimate focus on climate change today, there may be a risk of sub-optimizing AFS at the expense of other impacts, such as those categorized in Köthke et al. (2022) (Table 6), when suggesting changes in the AFS or comparing them to other systems. Thus, there is a need to consider other categories in the LCIA phase beyond climate change to avoid burden-shifting. To address this issue, some LCIA methods and characterization factors are available for impacts that are less frequently assessed, such as biodiversity (Knudsen et al., 2017; Koellner et al., 2013), biotic production potential (Hauschild et al., 2018), or erosion regulation (Hauschild et al., 2018), among others. However, there is also a challenge in capturing all the potential environmental effects of AFS, as discussed below.

The following sub-sections will discuss the environmental outcomes listed in Table 6 and follow a specific organization. First, the importance of the environmental outcome under consideration is shortly elaborated by referring to the broader agroforestry literature. Second, the methodological implications and state-of-the-art for capturing the environmental outcome in LCA are briefly discussed in the context of the general LCA literature. Finally, the selected studies are used as a reference to discuss if the environmental outcome under consideration is incorporated in the LCA (summary in Table 7).

Table 7
Frequency of environmental outcomes analyzed in the selected studies (n = 32), per AFS and in total. Dark color code: the number of studies considering this impact category is above the average of the total number of studies for all agroforestry groups.

Environmental outcome	Agrosilvicultural	Agrosilvopastoral	Silvopastoral	Total
Biodiversity	1	2	2	5
Climate change mitigation	4	6	5	15
Water	2	4	0	6
Soil	1	1	0	2
Pest and disease control	0	0	0	0
Pollination	0	0	0	0

4.2.1. Biodiversity

In the agroforestry literature, evidence on the effects of AFS on biodiversity varies. Some authors find that agroforestry positively affects biodiversity by creating habitats and connectivity for pollinators, birds, and important nitrogen-fixing species (Jose, 2009; Shibu, 2012; Udawatta et al., 2019). Other authors conclude that agroforestry has no beneficial effect on biodiversity (Mupepele et al., 2021) partly because exotic or invasive species are introduced (Ollinaho and Kröger, 2021; Schroth et al., 2004). In the LCA literature, while much evidence exists on biodiversity loss due to agriculture, evidence of biodiversity impacts in AFS is sparse. It is, therefore, important to include biodiversity assessments to provide statements specific to the context of the LCA. The lack of worldwide applicable biodiversity models for agroforestry that cover all biodiversity levels and go beyond species richness (e.g., functional diversity, ecosystem diversity) may explain its exclusion from LCAs (Winter et al., 2017). Nevertheless, some characterization factors may be used as proxy measures for biodiversity in European AFS expressed as a potentially disappeared fraction (Knudsen et al., 2017) or other approaches, as discussed in Gabel et al. (2016). In total, only 5 studies reported on biodiversity impacts which were most often measured at the endpoint level (Lamnatou et al., 2022; Paolotti et al., 2016; Parra-Paitan and Verburg, 2022) or as species diversity (Costa et al., 2018) or genetic diversity and naturalness index (Rocchi et al., 2019).

4.2.2. Climate change mitigation

Whether agroforestry is a carbon sink or source depends on various factors, including its structural composition, agroecological conditions, and soil characteristics, as indicated in the agroforestry literature (Dixon, 1995; Feliciano et al., 2018). In addition, long-term emissions may outweigh removals depending on modelling approaches and fate of the woody biomass (Garnett et al., 2017; Harrison et al., 2021). In the LCA literature, carbon sequestration is the main process that can be considered in the LCI phase for addressing climate change mitigation potential within the Climate Change impact category. Different approaches and tools are available for below- and above-ground biomass assessments in LCA (Burgess et al., 2019; Goglio et al., 2015; IPCC, 2006; Petersen et al., 2013), which may explain the great variability in the carbon footprint of AFS, as shown in Fig. 3. In total, 15 studies include inventory data on carbon sequestration (e.g., soil carbon stocks, allometric data of the woody biomass.), whereas the remaining studies solely consider GHG emission sources without accounting for potential carbon sinks. However, some studies account for carbon sequestration only in soils (Eldesouky et al., 2018; Escribano et al., 2022; Gutiérrez-Peña et al., 2019; Horrillo et al., 2021, 2020; Parra-Paitan and Verburg, 2022; Rowntree et al., 2020) or only in the woody biomass (Brook et al., 2022; Doddabasawa et al., 2020; Duffy et al., 2021; Mazzetto et al., 2023). A handful of studies include both, carbon sequestration in soils and the woody biomass (Crous-Duran et al., 2019; de Figueiredo et al., 2017; Martinelli et al., 2019; Reyes-Palomo et al., 2022). Furthermore, some studies do not differentiate the effects for all tree species (Raschio et al., 2018) or do not include carbon sequestration due to the lack of reliable data (Brook et al., 2022) or methodological difficulties (Costa et al., 2018). Other studies assume that soil carbon sequestration is in equilibrium, continues at the same rate indefinitely, or is estimated over a 10-year period (Brook et al., 2022; de Figueiredo et al., 2017; Rowntree et al., 2020). Thus, the utilization of different methodological approaches to estimate above- and below-ground carbon sequestration and the chosen time horizon can potentially impact the results, which can partly explain why some meat production systems have been reported as net carbon sinks in Fig. 3, despite their general carbon-intensive nature (Gaillac and Marbach, 2021).

4.2.3. Water

The broader agroforestry literature suggests that AFS can potentially improve groundwater quality and reduce nutrient leaching and encourage infiltration, sediment deposition, and nutrient retention (for example, in riparian buffers) (Jose, 2009; Silvestri et al., 2012). In the LCA framework, some of these aspects will be captured in the eutrophication impact category. With regard to water use, there has been substantial progress in methodological developments regarding water footprints in the LCA community (Berger and Finkbeiner, 2010; Gerbens-Leenes et al., 2021). In total, in six of the selected studies, water resource depletion (Acosta-Alba et al., 2020; Lamnatou et al., 2022), water footprint (Armengot et al., 2021; Caicedo-Vargas et al., 2022), net water use (Bianchi et al., 2021), and water scarcity (Costa et al., 2018) were included. However, none of the aforementioned methodologies, take into account the potential effect of AFS on water regulating services, retention, and infiltration linked to the soil hydrological properties, as described in Köthke et al. (2022).

4.2.4. Soil

According to the agroforestry literature, trees provide many physical, chemical, and biological benefits in soils by recycling nutrients from organic residues and increasing nutrient stocks over time (Jose, 2009). Nitrogen-fixing trees can improve soil quality (Jose, 2009) and microbiome populations (Kremer, 2021). Silvopastoral systems can positively impact soil fertility and erosion control (Torralba et al., 2016) by improving physical properties such as soil aggregation, porosity and pore connectivity (Rao et al., 1997). At the same time, management practices such as tillage and clear-cutting can lead to erosion (Kremer, 2021), and cattle can cause mechanical damage to crops (Atangana et al., 2014b). In LCA, it is crucial to acknowledge changes in chemical, biological, and physical soil properties (Goglio et al., 2018). However, there is no consensus on the best way to account for these changes in the LCA framework (Goglio et al., 2018). Of the selected studies, only two included data on soil health, such as soil compaction, soil organic carbon (used as a proxy for soil quality), soil erosion, or soil macro, meso, and microfauna (Costa et al., 2018; Utomo et al., 2016).

4.2.5. Pest and disease control

According to the agroforestry literature, biological pest control in AFS is possible due to improved prey-predator populations (Jose, 2009; Rao et al., 1997). Agroforestry can effectively reduce the pressure from pests and diseases, especially in perennial crops (Pumariño et al., 2015), while in annual crops, Pumariño et al. (2015) found no effect of AFS on pest abundance and plant damage. Modifications to tree canopy in agricultural systems can influence the microclimate (Atangana et al., 2014b) and provide shelter to pests or predators (Ango et al., 2014; Zhang et al., 2007), which may influence pesticide use (Zhang et al., 2007). In LCA, activity data related to pests and diseases is typically collected at the LCI phase (e.g., pesticide use), and the environmental effects are reported at the LCIA phase as toxicological or ecotoxicological impacts on human and ecosystems health (Dorca-Preda et al., 2022) which might partly capture the effect of AFS on disease prevalence or pest abundance. In total, none of the selected studies report data on disease prevalence and pest abundance or include it when reporting on ecotoxicity or toxicological impacts.

4.2.6. Pollination

Global evidence shows that pollination by insects can have positive or negative interactions in a given environment since pollination can occur with native or invasive plants (Gutierrez-Arellano and Mulligan, 2018). In the LCA framework, possible indicators that could be used to measure animal pollination from bees, bumblebees, wild bees, midges, or beetles are pollinator richness and pollen deposition or abundance, among others (Gutierrez-Arellano and Mulligan, 2018). However, there are currently no fully operational LCIA models that can include pollination due to insufficient data on species richness and geographical range (Crenna et al., 2017), which may explain why none of the selected studies examined pollination. Yet several pressures on insect pollination have been identified, leading to promising LCA frameworks and conceptual models that could incorporate pollination activities in the future (Crenna et al., 2017; Koellner et al., 2013; Zhang et al., 2010). Despite pollinators' essential role in AFS (Jose, 2009), only one of the selected studies mentioned its importance (Rowntree et al., 2020), yet none of the selected studies report on pollination in their LCA.

4.3. Limitations of the present review

There are a few limitations to bear in mind when interpreting the results of this paper. It is possible that the systematic literature search and the inclusion criteria did not capture all relevant publications addressing the research questions. The search strings might have missed relevant publications written in other languages. Also, many studies reporting on agroforestry were excluded because they did not consider food as the primary or secondary product in the LCA (see e.g., González-García et al., 2013; Dias et al., 2014). Other studies conducting a footprint analysis were excluded since they did not explicitly mention or apply the LCA framework (see e.g., Ortiz-Rodríguez et al., 2016; Rakotovao et al., 2021, 2017). Moreover, potential publication bias could also have influenced this paper, as studies reporting negative or less significant results might not have been published in the literature search. Furthermore, the screening and selection process was conducted by the lead author of this paper, which could potentially introduce bias. However, the risk and impact of bias were minimized by peer-checking and the use of pre-defined inclusion criteria and established guidelines and protocols, collectively contributing to the overall rigor of the study.

Moreover, it is difficult to categorize studies as 'agroforestry' – in the present review – since many definitions in the literature seldom align with each other (Ollinaho and Kröger, 2021). In addition, some studies do not explicitly state that their LCA is based on agroforestry, even though the structural components are assessed (Acosta-Alba et al., 2020). Other studies do not specify the type of AFS analyzed (Bianchi et al., 2021), include all the structural components of the system boundary (e.g., cattle, pigs, or poultry) (Caicedo-Vargas et al., 2022; Martinelli et al., 2019), or specify the type of husbandry or cereals analyzed (Acosta-Alba et al., 2020; Eldesouky et al., 2018). Consequently, what is agroforestry in one study may not be in another and may be subject to the authors' and this paper's own interpretation. Furthermore, pinpointing which environmental impacts are the *most* important to agroforestry is a challenging task in this paper since other 'non-mainstream' impacts across the agroforestry food value chain may be currently inadequately understood or methodologically challenging to operate in LCA (e.g., vibration and artificial light pollution). Thus, not all potentially relevant environmental outcomes have been discussed in this paper, and the gaps in knowledge may extend beyond the six environmental outcomes reported in Köthke et al. (2022). These shortcomings must be considered when analyzing this paper's results.

5. Conclusion

A total of 32 LCAs in 17 countries spanning a decade form the basis of this paper, which suggests that agroforestry occupies a niche in the literature of LCA. Around half of the selected studies are located in tropical climates, the rest being in temperate climates, predominantly in Southern Europe. The findings in this paper highlight the most common methodological approaches and provides the basis for future research on methodological issues and knowledge gaps regarding relevant environmental outcomes in the agroforestry literature that could be explored further in LCA.

The most common methodological approaches applied in the selected LCA studies are product-related functional units based on mass values and cradle-to-farm gate system boundaries. Most allocation methods are based on physical properties. Climate change has the greatest coverage from all impact categories with large variations across and within different production systems. These variations are, besides differences in production systems, partly influenced by methodological choices, such as handling of multifunctionality and applied method for above- and below-ground carbon sequestration, which may explain why some production systems that are commonly carbon-intensive, such as meat production, are reported as net carbon sinks.

The findings show that further methodological advancements are needed in LCA to include other environmental impact categories highlighted in the agroforestry literature beyond climate change such as the effects of AFS on biodiversity, carbon sequestration, water, and soil. This paper also shows important knowledge gaps in the LCA literature concerning different environmental outcomes in relation to different agroforestry systems in different locations. For instance, although the number of studies considering biodiversity is above the average of the total number of studies for all agroforestry groups, only one agrosilvicultural study reports about biodiversity. Likewise, while carbon sequestration is included in nearly half of the studies, the evidence for agrosilvicultural studies is limited. Gaps in knowledge can be attributed to a lack of unified operational indicators and impact categories within the LCA framework that are central in the general agroforestry literature.

Although LCA can partially capture some environmental outcomes, standardized approaches and further methodological improvements remain necessary to determine the net environmental effects of food products resulting from individual AFS, especially within the area of multifunctionality, carbon sequestration and biodiversity.

CRediT authorship contribution statement

The present study is a result of the collaborative efforts of all authors involved.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that this publication is free from any known conflicts of interest.

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Appendix A. Supplementary data

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A.2. Supplementary materials

Supplementary Material

How does Life Cycle Assessment capture the environmental impacts of agroforestry? A systematic review

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S-1. Review protocol

Based on Zumsteg et al. (2012) recommendations, the standardized technique for assessing and reporting reviews of LCA (STARR-LCA) was applied as shown in **Table S1.**

Table S1
STARR-LCA Worksheet for Researchers and Reviewers (Zumsteg *et al.*, 2012).

Item	Description	Included?[a]
1. Document title, structured summary, and key words	• Title identifies article as systematic review, meta-analysis or both • Abstract contains background; objectives; data sources; study eligibility criteria; scope, system boundaries and functional unit; study appraisal and synthesis methods; results; limitations; and conclusions with implications for key findings • Key words include meta-analysis and/or systematic review	Y Y Y
2. Rationale of the review	• Purpose of review study in the context of current knowledge	Y
3. Review questions and objectives	• Question elements consistent with PIFT format	Y
4. Description of review protocol	• How possible studies or data for review were located • Information sources • Description of electronic search strategies • Process for selecting studies or data to include in the review summary • Description of further analyses	Y Y Y Y Y
5. Findings and features of individual studies in the review	• Include major findings, methods and limitations • Present data graphically if possible	Y Y
6. Assessment of bias	• Assessment of bias for individual studies included and across studies when summarized • Statement of funding source for the review	N Y
7. Synthesis methods	• Description of how data was summarized qualitatively and quantitatively	Y
8. Limitations of the Review	• Limitations of methodology • Guidance about appropriate generalization or application of review findings	Y Y
9. Summary of findings and conclusions	• Clear conclusions • Discussion of conclusions in the context of other evidence	Y Y

[a]Y = yes; N = no; I = incomplete.

The preliminary data search (i.e., review scoping, **Table S2**) was initially broad with irrelevant studies of energy crops, forestry, timber, biofuel production, socio-economic assessments, and energy analysis. Thus, the definition of agroforestry interventions were expanded based on Nair (1993), Bilotta, Milner and Boyd (2014), and Eksvärd (2016). The systematic data search (including manual search) was conducted between the 21st of September 2021 and the 20th of September, 2022.

Table S2
Review scoping.

Date	Database	Search term	Hits
21/12/2021	Web of Science	TS=((LCA OR "Life Cycle Assessment" OR "Life Cycle Analysis" OR "environmental assessment" OR "environmental analysis" OR "carbon footprint" OR "environmental footprint" OR "Product Environmental Footprint" OR pef) AND (agroforestry OR "forest farm*" OR "farm woodland*" OR silvopast* OR silvoarable OR agr?silvopast* OR agr?silvicultur* OR "wood* pasture" OR hedgerow* OR windbreak* OR "riparian woodland*" OR "riparian buffer strip*" OR "buffer strip*" OR "riparian buffer*" OR "shelter belt*" OR "alley cropping" OR "tree system" OR "forest grazing"))	77
21/12/2021	Web of Science	TS=((LCA OR "Life Cycle Assessment" OR "Life Cycle Analysis" OR "environmental assessment" OR "environmental analysis" OR "carbon footprint" OR "environmental footprint" OR "Product Environmental Footprint" OR pef) AND (agroforestry OR "forest farm*" OR "farm woodland*" OR silvopast* OR silvoarable OR agr?silvopast* OR agr?silvicultur* OR "wood* pasture" OR hedgerow* OR windbreak* OR "riparian woodland*" OR "riparian buffer strip*" OR "buffer strip*" OR "riparian buffer*" OR "shelter belt*" OR "alley cropping" OR "tree system" OR "forest grazing") NOT (energy OR biofuel OR ethanol OR biomass))	32
22/12/2021	Web of Science	TS=((LCA OR "Life Cycle Assessment" OR "Life Cycle Analysis" OR "environmental assessment" OR "environmental analysis" OR "carbon footprint" OR "environmental footprint" OR "Product Environmental Footprint" OR pef) AND (agroforestry OR "forest farm*" OR "farm woodland*" OR "farm forest*" OR silvopast* OR silvoarable OR agr?silvopast* OR agr?silvicultur* OR "wood* pasture" OR hedgerow* OR windbreak* OR "riparian woodland*" OR "riparian buffer strip*" OR "buffer strip*" OR "riparian buffer*" OR "shelter belt*" OR "alley cropping" OR "tree system" OR "forest grazing" OR forestry OR "liv* fence*" OR "shade crops" OR "hillside system*" OR taungya OR "linear strip" OR "mixed forestry" OR hedge* OR shrub OR montado OR "home garden*" OR "tree garden*" OR "mixed wood" OR aquaforestry OR "agrosilvo fishery" OR "wood pasture*" OR "tree intercrop*" OR "grazed forest*" OR "woodland chickens" OR "food forestry" OR "protein banks" OR "apiculture with trees" OR "fodder trees"))	509

Several components were included in the development of the research question and used as the basis for developing the exclusion and inclusion criteria. Namely, product or process (P), impact(s) of interest (I), flows or economic sectors included (F), and type(s) of life cycle assessment (T) (**Table S3**). Due to the limited research on this topic, the focus is based on the global literature of AFS.

Table S3
PIFT components based on Zumsteg, Cooper and Noon (2012).

Product or process (P)	Impact(s) of interest (I)	Flows or economic sectors included (F)	Type(s) of life cycle assessment (T)
Global food products	All	Agroforestry	Process-based LCA

S-2. Data analysis

Table S4
Coding themes and aggregated categories.

Aggregated Category	First-order theme	Second-order theme	Third-order theme
(1) Agroforestry practices[a]	Agrosilvicultural	Woody perennials and arable crop production	-
	Agrosilvopastoral	Woody perennials, livestock, and arable crop production	-
	Silvopastoral	Woody perennials and livestock production	-
(2) Agroforestry components	Woody perennials (Trees[b]/Shrubs)	Shade trees	Inga Tree (*Inga sp.)*, lead tree (*Leucaena sp.*), leguminous tree (*Gliricidia sp.*), shade tree (species not specified)
		Fruit trees	Coconut tree (*Cocos nucifera*), plantain (*Musa sp.),* banana *(Musa sp.),* mango *(Mangifera indica),* Olives (*Olea europaea*), palm tree (species not specified or *Acrocomia aculeata)*, guaba, orange, chontaduro, fruit trees (species not specified), nut trees (species not specified or *Dipteryx alata, Anacardium occidentale*), mandarin (*Citrus reticulata*), orange (*Citrus sinensis),* avocado (*Persea americana),* guava (*Psidium guajava),* papaya (*Carica papaya),* wild mango (*Jacaratia spinosa),* Genipa (*Genipa americana),* cherries, apples, sorbs, plums, peaches, mulberries, pomegranates, figs and berries
		Non-timber trees	Rubber tree (*Hevea brasiliensis*), oak tree (holm oak), cork oak (*Quercus suber)*
		Timber trees	Eucalyptus tree (*Eucalyptus sp.*), teak (*Tectona grandis*), Neem (*Azadirachta indica*), Laure, Cedar
		Energy trees	Willow *(Salix spp.),* hazelnut tree, alder tree
		Other	Forest trees (species not specified), medicinal tree (guayusa), forest meadows, forest pasture, shrub pasture, elm tree, native forest (species not specified), Erythrina Fusca trees, Gliricidia trees, Erythrina Poeppigiana trees, Trichanthera Gigantea trees, Fig trees
	Animals[c]	Ruminants	Cattle, sheep, goat
		Monogastric	Pigs, rabbits
		Poultry	Chickens
		Other	Bees
	Crops	Food/feed crops	Soy, barley, maize, sorghum,

			wheat, pigeonpea, cassava, beans, cabbage (black/kohlrabi/savoy),nchards, Asparagus, artichokes, sweet potatoes, radishes, oats, vetch, barley, pea, cereals (crops not specified), forage crops (crops not specified)
		Cash crops	Coffee, sugarcane, cocoa
(3) LCIA (midpoint)[d]	Acidification	Acidification potential (ILCD, CML, ReCiPe, Eco Indicator-99, IMPACT World+)	-
	Eutrophication	Terrestrial eutrophication, freshwater eutrophication, marine eutrophication, eutrophication potential (ILCD, CML, ReCiPe, Eco Indicator-99, IMPACT World+)	-
	Resource depletion	Biotic and abiotic resource depletion (CML), net water use (ReCiPe), water/fossil/ozone layer depletion (ILCD, CML, ReCiPe), water scarcity (ILCD, Eco Indicator-99), and resource availability (ReCiPe), water footprint (CML)	-
	Climate Change	Global Warming Potential, Climate Change (ILCD, CML, ReCiPe, Eco Indicator-99, IPCC, IMPACT World+)	-
	Human toxicity	Cancer and non-cancer effects (USEtox, ILCD), toxicological potential (CML), human toxicity potential (IMPACT World+)	-
	Ecotoxicity	Ecotoxicity (USEtox, Eco-indicator 99) and terrestrial (CML, ReCiPe), marine aquatic (CML), and freshwater (ILCD, CML, UNEP-SETAC, ReCiPe, IMPACT World+)	-
	Human health	Respiratory organics and inorganics (Eco Indicator-99), ionizing radiation (ReCiPe), particulate matter (ILCD), carcinogens (Eco-indicator 99), radiation (Eco-indicator 99)	-
	Resource consumption	Non-renewable primary energy consumption	-

		(Impact 2002+)	
	Land use	Agricultural land occupation (ReCiPe), land use (Eco Indicator-99, UNEP-SETAC), natural land transformation (ReCiPe), land footprint (CML)	
	POC, PO or POF	Photochemical Oxidant Creation (ReCiPe), Photochemical Oxidation or Photochemical Ozone Formation (CML, ILCD)	-
	CED	Cumulative Energy Demand and non-renewable Cumulative Energy Demand (CML, ReCiPe)	-
(3) LCIA (endpoint)[d]	Human health	Respiratory organics and inorganics (Eco Indicator-99), carcinogens (Eco-indicator 99), radiation (Eco-indicator 99), human health (ReCiPe, Impact World+), ozone layer (Eco-indicator 99), climate change (Eco-indicator 99), disability-adjusted life years (Impact World+)	-
	Ecosystems	Ecosystem quality (Eco Indicator-99, Impact World+), ecosystem health (ReCiPe), ecotoxicity (Eco-indicator 99), acidification (Eco-indicator 99), eutrophication (Eco-indicator 99), land use (Eco-indicator 99)	-
	Resources	Resource consumption (Eco Indicator-99), Resources (ReCiPe), minerals (Eco Indicator-99), fossil fuels (Eco Indicator-99)	-
(3) LCIA (other)	Soil quality	Soil erosion, compactation, and SOC (AgBalance, laboratory analysis), soil microbes (laboratory analysis)	
	Water	Water footprint (Other[e])	
	Energy	Gross energy production (not specified), primary energy consumption (Other)[f]	
	Biodiversity	Biodiversity (AgBalance, GLOBIO-InVEST), genetic diversity index[g],	

		naturalness index[g]	
(4) Life cycle impact contributions	Direct/indirect field emissions	On-site farm emissions	Farm emissions (not specified), inputs application, enteric fermentation, and manure and/or soil management
	Raw material extraction	Off-farm emissions	Production of agrochemicals and seeds, fertilizers, raw materials (not specified), and off-farm feed
	Operation	Maintenance, fuel/electricity, or tools	Use of machinery, fossil fuels, and irrigation
	Manufacturing	Manufacturing	Processing and packaging
	Transportation	Transportation	Transport of goods or materials
	Does not mention	Does not mention	-

[a] Based on functional diversity (Nair, 1993).
[b] Based on primary function.
[c] Based on type of digestive system.
[d] Impact assessment method in parenthesis.
[e] Based on Hoekstra *et al.* (2011).
[f] Based on Saling *et al.* (2002).
[g] Based on Castellini *et al.* (2012).

Table S5
Data extraction template used in Covidence.

Theme	**Question**	**Response**
Basic information		
ID	What is the ID of this study?	Numeric value
Location(s)	Where is the study conducted?	Text
Year	What is the year of publication?	Numeric value
(1) LCA Phase: Goal and Scope		
Goal and scope	What is the objective of the study?	Text
Functional Unit (type)	Which functional unit(s) is used? Check all that apply	Mass/Economic/Area/Other
Functional Unit (metric)	What functional unit value(s) are used?	Numeric value(s)
System boundary (SB)	What is the system boundary? Check all that apply	Cradle-to-gate/Cradle-to-grave/Other
Type of agroforestry system	What type of AFS is used, based on the structural composition? Check all that apply	Agrosilvicultural/Agrosilvopastoral/ Silvopastoral/Other
Agroforestry components (tree)	What type of tree(s) are analyzed? Check all that apply	Shade trees/Fruit trees/Non-timber trees/Timber trees/Does not mention/ None/Other
Agroforestry components (crop)	What type of crop(s) are analyzed? Check all that apply	Food or feed crops/Cash crops/Does not mention/None/Other
Agroforestry components (animal)	What type of animal(s) are analyzed? Check all that apply	Ruminants/Monogastric/Poultry/Does not mention/None/Other
(2) LCA Phase: Life Cycle Inventory		
Multifunctionality	How is multifunctionality addressed? Check all that apply	System expansion/Economic allocation/Mass allocation/Does not mention/Other
(3) LCA Phase: LCIA		

Impact assessment model	Which impact assessment categories and/or models are included? Check all that apply	Midpoint/Endpoint/Other
Impact Categories	Which impact categories are included? Check all that apply	Acidification/Eutrophication/Resource depletion/Climate Change/Human toxicity/Ecotoxicity/Resource consumption/Land use/POC, PO, or POF/Cumulative Energy Demand/Human health/Ecosystems health/Resources/Other
(4) LCA Phase: Interpretation		
Results	What are the results of the Global Warming Potential (Climate Change) mass functional unit(s)? In kg CO_2eq/FU/year	Numerical value(s)
Sensitivity analysis	Does the study conduct a sensitivity analysis? Choose one option	Yes/No
Uncertainty analysis	Does the study conduct an uncertainty analysis? Choose one option	Yes/No
Contribution analysis	Does the study conduct a contribution analysis? Choose one option	Yes/No
Primary Impact contribution[a]	What is the primary impact contribution of Global Warming Potential (Climate Change)? Choose one option	Direct or indirect field emissions/Raw material extraction/ Operation/Manufacturing/ Transportation/Does not mention
Secondary Impact contribution[a]	What is the secondary impact contribution for Global Warming Potential (Climate Change)? Choose one option	Direct or indirect field emissions/Raw material extraction/ Operation/Manufacturing/ Transportation/Does not mention

[a] Descriptions of each impact contribution are elaborated in **Table S4**.

S-3. Results

Table S6
Extended table of selected LCA studies.

ID	Source	Location	Type of AFS	Goal and Scope	Primary output	Functional Unit (FU)	System Boundary	Impact Category[a]	Statement of funding source
1	Acosta-Alba *et al.* (2020)	Colombia	Agrosilvicultural Agrosilvopastoral	To compare three types of coffee cropping systems representative of Colombian coffee farming	Coffee	Mass Area Economic	Cradle-to-gate	CC, EU, AC, RD, and EC	Yes
2	Armengot *et al.* (2021)	Bolivia	Agrosilvicultural	To compare the food-energy-water nexus of four cacao systems	Cocoa	Mass	Cradle-to-gate	CC, EU, AC, RD, EC, and POC, PO or POF	Yes
3	Bianchi *et al.* (2021)	Indonesia Ghana Ecuador	Agrosilvicultural	To assess the environmental impacts of dark, milk and white chocolate	Chocolate	Mass Energy	Cradle-to-grave	CC, EU, AC, RD, CED, and POC, PO or POF	Yes
4	Brook *et al.* (2022)	Costa Rica	Silvopastoral	To calculate total greenhouse gas (GHG) emissions from dairy farms in 2016, 2017, and 2018	Milk	Mass	Cradle-to-gate	CC	Yes
5	Caicedo-Vargas *et al.* (2022)	Ecuador	Agrosilvopastoral	To compare the environmental and economic performance of cacao under conventional and organic agroforestry production systems	Cocoa Total harvested crops	Mass Area	Cradle-to-gate	CC, EU, AC, RD, CED, EC, HT, and POC, PO or POF	Yes
6	Caputo *et al.* (2020)	Italy	Agrosilvicultural	To evaluate the sustainability of peri-urban agriculture projects (five start-ups)	Total harvested crops	Mass Area	Cradle-to-grave	CC and RC	Yes

				for urban regeneration					
7	Costa *et al.* (2018)	Brazil	Agrosilvopastoral	To evaluate the socio-eco-efficiency of five different combinations of crops, forestry, and cattle systems	Total energy supply of edible products and energy sources	Composite	Cradle-to-gate	CC, EU, AC, RD, EC, LU, RC, and POC, PO or POF	Yes
8	Crous-Duran *et al.* (2019)	Portugal	Agrosilvicultural	To estimate the GHG emissions of crop and tree production and carbon sequestration (C-seq)	Wheat	Mass	Cradle-to-gate	CC	Yes
9	de Figueiredo *et al.* (2017)	Brazil	Agrosilvopastoral	To estimate the carbon footprint (CF) of beef cattle in three production scenarios	Cattle	Mass	Cradle-to-gate	CC	Yes
10	Doddabasawa *et al.* (2020)	India	Agrosilvicultural	To assess the CF in agroforestry systems that are rainfed and irrigated	Pigeonpea	Area	Cradle-to-gate	CC	No
11	Duffy *et al.* (2021)	Costa Rica	Silvopastoral	To analyze the efficacy and economic efficiency of potential GHG mitigation measures for tropical and subtropical dairy systems	Milk	Mass	Cradle-to-gate	CC	Yes
12	Eldesouky *et al.* (2018)	Spain	Agrosilvopastoral	To study the CF of Dehesa agroforestry systems	Sheep Cattle Milk	Mass	Cradle-to-gate	CC	Yes
13	Escribano *et al.* (2022)	Spain	Agrosilvopastoral	To analyze the technical-economic management and evaluate the CF and C-seq from organic livestock	Total sold animal	Mass Area	Cradle-to-gate	CC	Yes
14	Gutiérrez-Peña *et al.*	Spain	Agrosilvopastoral	To analyze the CF of grazing dairy goat	Goat	Mass	Cradle-to-gate	CC	Yes

	(2019)			systems according to their grazing level					
15	Horrillo *et al.* (2021)	Spain	Agrosilvopastoral	To estimate the maximum Carbon price of organic livestock farms in Dehesas and rangelands	Beef Sheep Pig	Area	Cradle-to-gate	CC	Yes
16	Horrillo *et al.* (2020)	Spain	Agrosilvopastoral	To estimate the CF and C-seq in seven ruminants and Iberian pig organic farms	Beef Sheep Pig Goat	Mass Area	Cradle-to-gate	CC	Yes
17	Lamnatou *et al.* (2022)	Spain	Agrosilvopastoral	To evaluate the environmental performance of Iberian-pig production system in the growing-fattening stage	Pig	Mass	Cradle-to-gate	CC, EU, RD, AC, HT, EC, CED, LU HH, E, R, POC, PO, or POF	Yes
18	Lehmann *et al.* (2020)	Brazil Italy	Silvopastoral Agrosilvicultural	To compare four agroforestry systems with olive production in Italy and combined food and energy system and conventional in Denmark	Olives	Mass Area Economic	Cradle-to-gate	CC, EU, AC	Yes
19	Livingstone *et al.* (2021)	Ireland	Silvopastoral	To analyze the environmental impacts on food, energy and water of Short Rotation Coppice willow riparian buffer strips in intensive agricultural applications	Farmland	Area	Cradle-to-gate	CC, EU, GEP	Yes
20	Martinelli *et al.* (2019)	Brazil	Agrosilvicultural	To assess the global warming potential (GWP) of five agroforestry systems, based on their capacity	Agroforestry area	Area	Cradle-to-gate	CC	Yes

				for storing carbon					
21	Mazzetto *et al.* (2020)	New Zealand	Silvopastoral	To estimate the CF of beef and sheep meat including overseas markets	Beef Sheep	Mass	Cradle-to-grave[b]	CC	Yes
22	Paolotti *et al.* (2016)	Italy	Silvopastoral	To analyze the environmental impact of integrated free-range poultry and olive orchards	Poultry	Mass	Cradle-to-gate	HH, E, R	Yes
23	Parra-Paitan and Verburg (2022)	Ghana	Agrosilvicultural	To calculate the impacts caused by cocoa production at the farm-level	Cocoa	Mass	Cradle-to-gate	CC, EU, EC, AC, HT, E, HH	Yes
24	Pérez-Neira *et al.* (2020)	Ecuador	Agrosilvicultural	To quantify the environmental impact of dark chocolate (100% cacao), including the production, manufacture, and transportation phases until retail	Chocolate	Mass	Cradle-to-retail	CC, EU, RD, AC, HT, EC, LU, CED, and POC, PO or POF	Yes
25	Raschio *et al.* (2018)	Peru	Agrosilvicultural	To identify GHG emissions from cocoa farms, including spatiotemporal assessment of perennial crops	Cocoa	Mass Area	Cradle-to-gate	CC	Yes
26	Reyes-Palomo *et al.* (2022)	Spain	Silvopastoral	To calculate the CF of organic and conventional cattle Dehesa farms and C-seq	Cattle	Mass	Cradle-to-gate	CC	Yes
27	Ripoll-Bosch *et al.* (2013)	Spain	Agrosilvopastoral	To assess the GHG emissions of three meat-sheep farming	Lamb	Mass	Cradle-to-gate	CC	Yes

				systems					
28	Rocchi *et al.* (2019)	Italy	Silvopastoral	To assess the sustainability of three poultry production systems	Poultry	Mass	Cradle-to-gate	HH, E, R	Yes
29	Rowntree *et al.* (2020)	USA	Silvopastoral	To conduct a whole-farm LCA of a multi-species pastured livestock system converted from degraded cropland	Animal carcass weight	Mass	Cradle-to-gate	CC	Yes
30	Ruiz-Llontop *et al.* (2022)	Peru	Silvopastoral	To estimate the CF of milk production in dairy farms	Milk	Mass	Cradle-to-gate	CC	Yes
31	Tziolas *et al.* (2022)	Greece	Agrosilvicultural	To conduct a holistic environmental and economic assessment of agroforestry	Arable land	Area	Cradle-to-gate	CC	Yes
32	Utomo *et al.* (2016)	Indonesia	Agrosilvicultural	To evaluate the environmental performance of cocoa monoculture and cocoa-agroforestry	Cocoa	Mass	Cradle-to-gate	CC, AC, EU	Yes

[a] CC= Climate Change; AC= Acidification; EU= Eutrophication; RD= Resource Depletion; RC= Resource Consumption; HT= Human Toxicity; EC= Ecotoxicity; CED= Cumulative Energy Demand; GEP= Gross Energy Production; LU= Land Use; POC, PO, or POF= Photochemical Oxidant Creation, Photochemical Oxidation or Photochemical Ozone Formation; HH= Human Health; E= Ecosystems; R= Resources.
[b] Reports cradle-to-farm gate data separately.

Table S7
Extracted values for the Climate Change impact category in kg CO2eq FU^{-1} organized in three contrasting production systems (according to their primary output).

ID	Source	Case study[a]	Climate change[b] (kg CO2eq FU^{-1})	Functional Unit (FU)
Crop production system				
1	Acosta-Alba et al. (2020)	1) Coffee alone 2) Coffee with transition shade 3) Coffee with permanent shade	2) 4.6 3) 3.1	ton of parchment coffee
2	Armengot et al. (2021)	1) Organic agroforestry 2) Conventional agroforestry 3) Organic monoculture 4) Conventional monoculture	1) 1.56 2) 3.74	kilograms of cacao output
5	Caicedo-Vargas et al. (2022)	1) Organic agroforestry 2) Conventional agroforestry	1) 0.034 2) 0.300	kg of cacao
18	Lehmann et al. (2020)	1) Silvopastoral 2) Organic agroforestry 3) Traditional agroforestry 4) Conventional agroforestry 5) Agroforestry combined food and energy system 6) Conventional olive system	1) 0.166 2) 0.266 3) 0.655 4) 0.388	kg of olive yield
23	Parra-Paitan and Verburg (2022)	1) Cocoa agroforestry (3 scenarios) 2) Cocoa full sun	1) -0.03 (Scenario 1) 0 (Scenario 2) 0.06 (Scenario 3)	kg of cocoa beans ready for further processing
32	Utomo et al. (2016)	1) Cocoa monoculture 2) Cocoa-coconut agroforestry 3) Cocoa-rubber agroforestry	2) 3.67E+01 3) 7.65E+01	metric ton of cocoa pod
Milk production system				
4	Brook et al. (2022)	1) Dairy farm (3 scenarios)	1) 1.03 [0.75±0.25] (year 1) 1.14 [0.82±0.27] (year 2) 1.13 [0.84±0.26] (year 3)	kg FPCM
11	Duffy et al. (2021)	1) Specialized Dairy Extensive in the Lowland based on a 20% of farm area afforested 2) Specialized Dairy Intensive Lowland based on a 20% of farm area afforested 3) Specialized Dairy Semi-intensive in the Uplands based on a 20% of farm area afforested 4) Specialized Dairy Intensive in the Uplands based on a 20% of farm area afforested	1) 0.9 [c] 2) 1.1 [c] 3) 1.1 [c] 4) 0.9 [c] 5) 0.5 [c]	kg FPCM

		5) Dual-purpose Extensive in the Lowlands based on a 20% of farm area afforested		
14	Gutiérrez-Peña et al. (2019)	1) Agroforestry low productivity grazing farms 2) Agroforestry more intensified grazing farms 3) Agroforestry high productivity grazing farms	1) [2.36±0.32] (Milk correction 1) [1.40±0.19] (Milk correction 2) 2) [1.97±0.11] (Milk correction 1) [1.16±0.06] (Milk correction 2) 3) [1.76±0.13] (Milk correction 1) 1.04±0.08] (Milk correction 2)	kg FPCM
30	Ruiz-Llontop et al. (2022)	1) Dairy farm 54 ha 2) Dairy farm 44 ha 3) Dairy farm 88.6 ha 4) Dairy farm 22,5 ha 5) Dairy farm 22,5 ha 6) Dairy farm 29,0 ha 7) Dairy farm 90,0 ha 8) Dairy farm 53,5 ha	1) 1.9 2) 2.2 3) 3.09 4) 2.59 5) 1.77 6) 1.76 7) 2.08 8) 2.71	kg FPCM
16	Horrillo et al. (2020)	1) Agroforestry beef cattle calves 2) Agroforestry beef cattle yearlings 3) Rangeland meat sheep lambs weighting 23 kg 4) Rangeland meat sheep lambs weighting 18.5 kg 5) Agroforestry Iberian pig farm using the Montanera fattening system 6) Agroforestry Iberian pig closed herd 7) Agroforestry semi-extensive dairy goat	7) 1.19 [0]	kg FPCM in dairy farms
Meat production system				
9	de Figueiredo et al. (2017)	1) Degraded pasture 2) Managed pasture 3) Crop-livestock-forest integrated system	3) -28.1	kg LW
12	Eldesouky et al. (2018)	1) Extensive meat sheep farm 2) Extensive beef/calf cattle farm 3) Extensive beef/calf cattle farm with feedlot finishing of calves 4) Grazing dairy sheep farm	2) 17.74 [12.35][c] 3) 8.62 [6.34] [c]	kg LW of product (lambs or calves)
13	Escribano et al. (2022)	1) Agroforestry farms mainly producing organic cattle 2) Agroforestry mixed farms 3) Agroforestry largely intensified small farms	1) 18.04 (6.02)[c]	kg LW of sold animal (beef)
16	Horrillo et al. (2020)	1) Agroforestry beef cattle calves 2) Agroforestry beef cattle yearlings 3) Rangeland meat sheep	1) 16,27 [10.52] 2) 10,43 [5.25] 5) 2,94 [-3.58] 6) 4,16 [-2.15]	kg LW per sold animal (in meat farms)

		lambs weighting 23 kg 4) Rangeland meat sheep lambs weighting 18.5 kg 5) Agroforestry Iberian pig farm using the Montanera fattening system 6) Agroforestry Iberian pig closed herd 7) Agroforestry semi-extensive dairy goat		
17	Lamnatou et al. (2022)	1) Agroforestry extensive (growing-fattening) Iberian-pig system	1) 4.37 to 6.19	kg LW or carcass weight
21	Mazzetto et al. (2020)	1) Beef production 2) Sheep production 3) Traditional beef production	1) 8.97 [7.06] 2) 6.01 [4.26] 3) 10.09 [7.16]	kg LW
26	Reyes-Palomo et al. (2022)	1) Organic beef agroforestry 2) Conventional beef agroforestry	1) [0.9 ± 22.99] 2) [10.11 ± 20.91]	kg LW of calf at the end of the fattening period
27	Ripoll-Bosch et al. (2013)	1) Pasture-based agroforestry 2) Mixed agroforestry 3) Zero grazing	1) 25.9 (Scenario 1) 13.9 (Scenario 2) 2) 24.0 (Scenario 1) 17.7 (Scenario 2)	kg LW, leaving the farm-gate

[a] Case study in the selected study, including agroforestry and non-agroforestry.
[b] Values are only extracted for the case studies representing agroforestry systems. Values with C-seq are shown in brackets, and scenarios are in parentheses.
[c] Estimated values based on the reported data in the selected study of agroforestry systems.

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Appendix B

B.1. Paper II

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Research article

Life cycle assessment and modeling approaches in silvopastoral systems: A case study of egg production integrated in an organic apple orchard

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A B S T R A C T

This paper aimed to assess the environmental impacts of two organic silvopastoral farms in Austria, using a Life Cycle Assessment approach. The two farms (F1, F2), with egg production integrated into an apple orchard, were compared to standard practices for each product. The functional unit was '1 kg fresh Class I apples' and '1 kg fresh Class I eggs'. The assessment covered two scopes: cradle-to-farm gate and cradle-to-retail for each product. Effects on climate (including carbon sequestration in the soil and woody biomass), eutrophication potential (EP), acidification potential (AP), and land occupation (LO) were assessed. Feed, manure, and land were three resource loops included in the system boundary. Two modeling approaches were used from cradle-to-farm gate for distributing the impacts of the entire system between apples and eggs: model 1 (M1) used economic allocation, while model 2 (M2) divided the system into two subsystems. Results varied considerably by model. M1 consistently showed higher impacts for apples and considerably lower for eggs compared to M2. At farm gate, the carbon footprint (CF) ranged from 0.09 to 0.17 kg CO_2-eq/kg apple and 0.19–1.62 kg CO_2-eq/kg egg across all analyzed systems and models. Carbon sequestration reduced emissions by 22–42% for apples and by 0.4–39% for eggs. Sequestration was mainly associated with the carbon contributions from plant biomass from apple production (84–99%), with manure contributing 0.7–9%. EP ranged from 0.19 to 1.7 g PO_4-eq/kg apple and 0.7–35 g PO_4-eq/kg egg and AP ranged from 0.8 to 2.9 g SO_2-eq/kg apple and 2–36 g SO_2-eq/kg egg across all analyzed systems and models. LO ranged from 0.3 to 0.6 m^2/kg apple and 0.8–9 m^2/kg egg across all analyzed systems and models. Post-harvest activities accounted for up to 29% of the total impacts for EP and AP, and up to 57% for CF from cradle-to-retail. In general, the impacts per kg egg or kg apple in F1 and F2 were lower in most impact categories relative to their reference systems, driven mainly by management factors and the production phase of the value chain. Further development of modeling approaches is needed.

1. Introduction

Agriculture is a cornerstone of global livelihoods, playing a fundamental role in global food security (Ahmed et al., 2020). Perennial plantations, such as apple orchards, not only help sustain food production but also bring environmental benefits, including carbon (C) storage in the soil and plant biomass (Goossens et al., 2017). However, these systems can still place environmental pressure through the intensive use of fertilizers, herbicides, and pesticides (Bessou et al., 2013). Similarly, while well-managed animal production systems can enhance soil fertility by improving nutrient cycling through manure application (Gliessman, 2014), they are also responsible for 14.5% of global anthropogenic greenhouse gas (GHG) emissions (Ahmed et al., 2020). Egg production, more specifically, while less land-intensive than ruminant systems, has its own environmental challenges, particularly through the demands of feed production and manure management (van Hal et al., 2019).

In Austria, agriculture is responsible for 9.5% of the total national emissions, primarily due to methane and nitrous oxides (Anderl et al., 2022). Among the key agricultural activities, apple and egg production are important components in the sector (Geßl, 2020). Apples are a major fruit crop, with Austria producing approximately 200.000 tons annually,

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much of it from organic orchards, with 76% of the total production concentrated in the Styria region (Muder et al., 2022). Similarly, organic egg production has expanded significantly, and Austria is now one of the leading countries in Europe in terms of free-range production (Augère-Granier, 2019). However, the environmental footprint within these systems needs to be addressed (Zaller et al., 2023).

Food from agroforestry systems (AFS) is a potential practice that can address the growing concerns over environmental degradation. Silvopastures are examples of AFS, where the integration of egg production in fruit orchards has the potential to close several resource loops beneficial for the environment. In Europe, the combination of fruit trees and animals is the most widely adopted form of agroforestry (accounting for 68% of the total AFS involving fruit production). However, this practice remains limited in scale compared to conventional orchards, representing only 0.24% of the total utilized agricultural area and 6% of the total area planted with fruit trees (Dodds et al., 2019). Despite its limited extent, these systems can potentially improve nutrient cycling and increase soil organic matter (Sollen-Norrlin et al., 2020). Additionally, hens can derive supplementary feed from foraging, potentially reducing the intake of concentrates (Paolotti et al., 2016; Bosshardt et al., 2022). Furthermore, multiple outputs can be delivered from the same unit of land, reducing land occupation (Escribano et al., 2022). However, integrating hens into perennial crop systems presents complex environmental interactions. For example, soils can be damaged due to pecking, scratching, overgrazing, and soiling (Sales-Baptista et al., 2016; Bosshardt et al., 2022). Additionally, nitrogen hotspots often accumulate in areas surrounding the hen houses, where manure deposition is concentrated (Bosshardt et al., 2022).

Modeling the above-mentioned environmental interactions can be a complex endeavor, and various scientific methods, levels of analysis, and indicators can be used to assess the environmental impacts of AFS. Among these methods, Life Cycle Assessment (LCA) is a comprehensive tool that quantifies environmental impacts from a systems thinking perspective, encompassing the entire life cycle of a product or service (ISO, 2006a; Arvanitoyannis, 2008). Although LCA has been applied to AFS (Paolotti et al., 2016; Recanati et al., 2018; Lehmann et al., 2020; Ma et al., 2022), modeling silvopastoral systems remains a niche in the LCA literature and results vary greatly because of the different approaches used.

A recent review summarized some predominant methodological choices commonly used in LCAs of AFS (Quevedo-Cascante et al., 2023). For example, the C-seq potential is often omitted or included in very different ways, introducing difficulties in comparing results. In silvopastoral systems, C-seq is rarely accounted for in LCAs, and when included, the focus is often on soil, while C stored in woody biomass is often overlooked, highlighting a significant knowledge gap in LCAs of perennial systems (see e.g., Paolotti et al., 2016; Rosati et al., 2016; Tziolas et al., 2022). Only a handful of silvopastoral studies regarding dairy production (Brook et al., 2022) and beef and sheep meat (Mazzetto et al., 2023) address C-seq potential in the woody biomass, with reported offsets ranging from 26 to 28% (see eg., Brook et al., 2022). In contrast, soil C-seq has been more frequently considered in silvopastoral studies involving cattle production (Reyes-Palomo et al., 2022), dairy goat production (Gutiérrez-Peña et al., 2019), and multiple livestock products (e.g., meat sheep, dairy goat, Iberian ham) (Horrillo et al., 2020), with reported emissions compensations between 35 and 100% (see e.g., Horrillo et al., 2020). Moreover, no LCAs involving egg or apple production have been found to account for C-seq, underscoring another important gap in the literature. Furthermore, existing LCA studies usually do not extend beyond farm gate assessment (see e.g., Escribano et al., 2018) and only a handful was found to conduct a comparative analysis of their standard system (Paolotti et al., 2016; Utomo et al., 2016). Thus, limiting the scope of the analysis of the differences between products from standard systems and AFS. Moreover, studies predominantly use a 'black box' approach to handle the multifunctionality of integrative systems (Mazzetto et al., 2023). Thus, internal processes are not explicitly partitioned to specific farm outputs, suggesting a gap in knowledge for testing other approaches.

In response to such predominantly applied approaches in LCA studies, this paper has two objectives. First, to assess the environmental impacts and carbon footprint of two silvopastoral case studies in Austria, which involve egg production integrated into a perennial system (i.e., apple orchards), including a contribution analysis and considerations of C-seq and post-harvest activities. Second, to test two modeling approaches for handling multifunctionality at farm gate and compare the impact of the products from the assessed case studies to what is considered for each product, their standard and specialized practice.

2. Materials and methods

2.1. System description

The systems under analysis were based on two real-world silvopastoral farms located in Austria, in the Styria and Lower Austria provinces. This region is known for its significant apple production, featuring sandy and silty soils with an average pH of 6.5 and mean annual temperatures and precipitation levels of 10.6 °C and 800 mm, respectively. A map of the study area and the general silvopastoral configuration of the farms are shown in Fig. 1.

Two silvopastoral farms, Farm 1 (F1) and Farm 2 (F2), were purposively sampled (Phelan, 2011) because of their agroforestry activities, food-related products, farmers willingness to cooperate, and the availability and accessibility of empirical data. In addition, the two farms were sampled based on four key contrasting management practices, such as animal sourcing, animal density, replanting method, and between-row management (BRM). The selected farms were part of a larger network within the international European MIXED[1] project and relatively new in the agroforestry domain, having only recently incorporated animals (i.e., egg-laying hens and roosters) into their apple orchards. Farms had a well-established grass clover area and used drip tubing for irrigation. Compost was stored in windrows with infrequent turning and applied to both the agroforestry and non-agroforestry areas of the farmers' land. The duration and type of field operation were similar between the case studies and the reference system (per unit of area). All bird houses, built with similar materials, feature small wooden sheds with plastic grids, wooden perches, nests, windows, and round metal feeders and drinkers. Post-harvest activities were the same for all systems as they all followed organic farming principles. Thus, the systems were characterized by short value chains and distinguished between first-graded products (i.e., those that remained fresh for consumers, hereafter referred as to class I) and second-graded products (i.e., those that did not meet the market demands regarding size and esthetics and were transformed into processed products, hereafter referred as to class II). Technical characteristics are furthered summarized in Table 1.

- F1 had low stocking densities of hens in the orchard. F1 used spent hens (i.e., animals, bought from an organic farm after their first laying period) instead of pullets. After use, spent hens were either consumed at the farm or sent to a rendering plant at the end of their production cycle. Furthermore, trees were regrafted with minimal soil disturbance, leaving fine and coarse roots in the soil. Biomass from pruning was partially used for mulching, and compost was mixed with manure and applied every five years. Animals were fed ad libitum with concentrates (i.e., mainly protein) and grains were supplemented from arable land (Table S2 in SD).

[1] MIXED is a project funded by the EU Horizon 2020 program and involves 14 networks of organic and conventional farmers across Europe (https://projects.au.dk/mixed/about-mixed).

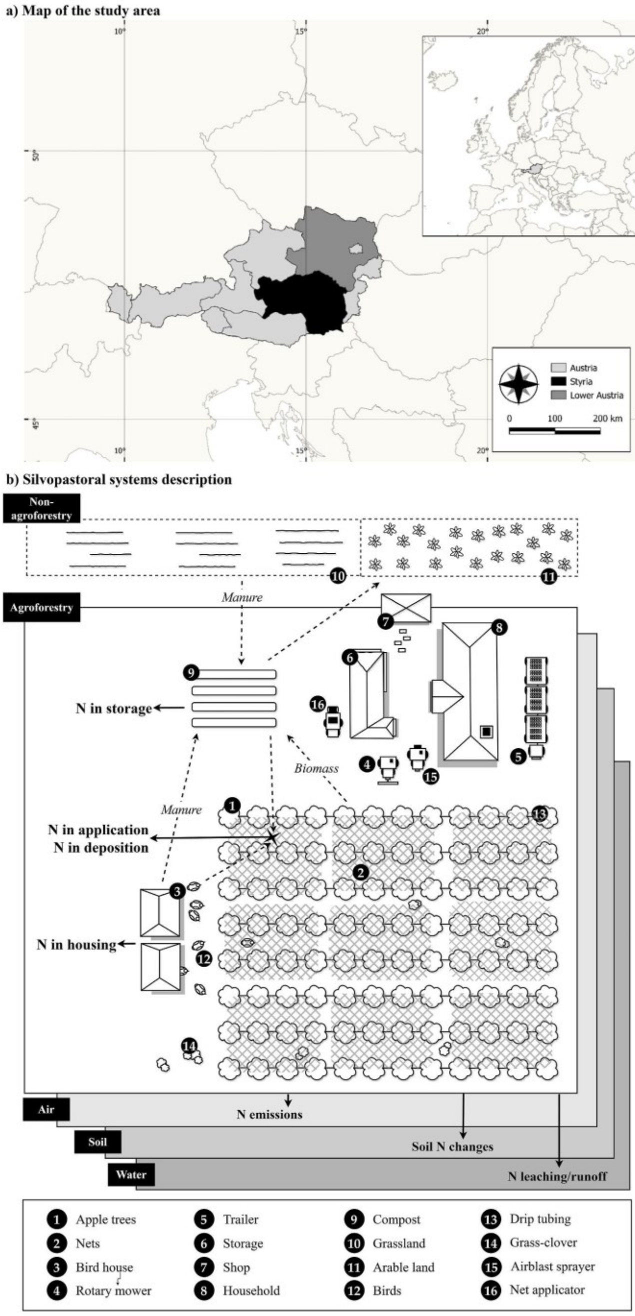

Fig. 1. a) map of the study area. b) general farm structure and components (not at scale), including N flows across three environmental compartments (air, water, and soil).

3

Table 1
Technical characteristics of the systems under study were obtained from surveys for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E). Key management differences are indicated in bold.

Characteristics	Unit	F1	F2	RS-A	RS-E
General					
Land	ha	9	9.9	1	2.85
Average annual precipitation	mm/yr	800	880	840	840
Average annual temperature	°C	11	9	10	10
Production	system	Organic and biodynamic	Organic and biodynamic	Organic	Organic
Apple system					
Cultivars	Type	Topaz, Gala	Topaz, Gala	Topaz, Gala	–
Tree density	#/ha	2400	2400	2400	–
Apple yield	kg FM/ha/yr	30000	30000	30000	–
Irrigation system	Type	Drip irrigation	Drip irrigation	Drip irrigation	–
Fertilization[a]	kg N/yr	6	52	47	0
Between-row-management	**type**	**Mulching**	**Mowing**	**Mowing**	–
Replanting method	**type**	**Grafting**	**Uprooting**	**Uprooting**	–
Harvest method	type	Manual and mechanical	Manual and mechanical	Manual and mechanical	–
Egg system					
Total flocks	#	2	5	–	1
Breed	type	Traditional local	Lohman Brown	–	Lohman Brown
Egg yield	eggs/hen/yr	180	293	–	293
Egg laying performance	%	78–40	85–60		85–60
Annual bird population	**#**	**116**	**877**	**-**	**2828**
Hens (25–56 weeks)	#	0	699	–	2268
Pullets (18–24 weeks)	#	0	150	–	560
Spent hens (70 weeks)	#	112	0		0
Roosters (25–56 weeks)	#	4	28		0
Feed intake, total[b]	g DM/bird/day	90	96	–	121
Concentrates[c]	g DM/bird/day	84	90	–	115
Forage[d]	g DM/bird/day	6	6	–	6
Age entering the farm	weeks	70	18	–	18
Animal input	**type**	**Spent hens**	**Pullets**	**-**	**Pullets**
Average period in the farm	weeks	52	56	–	56
Mortality	%	40	20	–	20
Egg losses	%	1	5	–	5
Flock distribution	type	Mixed	Mixed	–	Mixed
Average egg weight	g/egg	60	62	–	62
Houses	#	2	5	–	1[e]
Manure management	type	Pit storage	Pit storage	–	Pit storage
Artificial light	hrs/house/day	4	16	–	16

[a] N related to the applied compost, excluding N inputs from manure deposition.
[b] Detailed information regarding feed intake during the husbandry stage is in Table S2 in SD, including ingredients produced at the non-agroforestry farm level.
[c] All concentrates provided during the husbandry stage are imported to the agroforestry farm area.
[d] All foraging activities happen within the agroforestry farm area.
[e] Assuming one large building for all animals (equivalent to the materials and energy needed for 24 houses).

- F2 had a larger animal density in the orchard and almost double the egg yield per hen compared with F1. F2 used pullets for the replacement of hens at the end of their production cycle when the performance dropped to approximately 60%. Hens that reached the end of their production cycle were consumed or rendered. Animals received concentrates which was adjusted to cover the protein-rich feed requirements during winter and summer. Additionally, the feed was blended with grains harvested from the farmer's arable land (Table S2 in SD). Furthermore, after the end of the tree production cycle, deep plowing and coarse root excavation were undertaken for replanting new trees. All biomass from pruning was composted and mixed with sheep and bird manure. Compost was applied at least once a year.
- Reference systems: Two hypothetical farms associated with standard practices of non-integrated organic apples and egg production, respectively, were modeled using literature data and expert opinion. The reference system for apples (RS-A) was characterized by apple yields, tree densities, replating methods, and BRM identical to those of F2. The specialized egg (RS-E) system was characterized by the same egg output as F2, and birds were fed complete concentrates (Table S2 in SD).

2.2. Goal and scope definition

This study followed the four methodological steps of LCA (ISO, 2006a, 2006b), namely, goal and scope definition, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA), and interpretation. The goal of this study was twofold. First, to assess the environmental impacts of two silvopastoral farms that exemplify an integrated apple and egg production system, along with respective value chains and their C-seq potential, using a comparative and attributional[2] LCA methodology. Second, to test two modeling approaches for handling multifunctionality at the farm gate and compare the case studies against their respective reference systems. The rationale for testing these models lies in their distinct approaches to addressing multifunctionality in integrated systems. Model 1 (M1), which employs a 'black box' approach, is the most commonly used approach in LCAs of AFS (Quevedo-Cascante et al., 2023). To provide a complementary perspective aligned with the higher ISO hierarchy for dealing with multifunctionality (ISO, 2006a), Model 2 (M2) was used, where the farm system is subdivided into two subsystems. Both models are further detailed in **section 2.3.1.1**. A product-based functional unit (FU) was used from cradle-to-farm gate

[2] A "process based modelling intended to provide a static representation of average conditions, excluding market-mediated effects" (FAO, 2020).

Table 2
Annual input and output data at the farm level related to the production of apples and eggs in Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E). The table includes the applied allocation factors and carbon contributions for estimating sequestration in the soil and the woody biomass.

Input	Unit[a]	F1	F2	RS-A	RS-E
Animals	kg LW	209	1313	–	4188
Concentrates	kg	4051	32545	–	135282
Houses	p	0.1	0.3	–	1.6
Electricity	kWh/house	10	97	–	456
Straw (litter)	kg FM	365	456	–	2150
Water	m^3	7	56	–	233
Compost	kg N/ha	6	52	41	0
Phosphorous	kg P/ha	0.2	0.4	0.3	2.6
Irrigation	m^3/ha	60	70	70	–
Plant protection	kg/ha	1.5	1.5	1.5	–
Planting and establishment	p	0.1	0.1	0.1	–
Field operations					
Harvest	hrs/ha	27	27	27	–
Pruning	hrs/ha	27	27	27	–
Between-row-management	hrs/ha	3	13	13	–
Plant protection	hrs/ha	23	23	23	–
Fertilization	hrs/ha	0.1	1	1	–
Maintenance	hrs/ha	5	5	5	–
Fertilization	hrs/ha	1	1	1	–
Uprooting	hrs/ha	0	1	1	–
Output					
Apple (total)	kg FM/yr	270000	297000	30000	–
Class I	kg FM/yr	237600	261360	26400	–
Class II	kg FM/yr	32400	35640	3600	–
Egg (total)	kg FM/yr	1200	14615	–	48718
Class I	kg FM/yr	1080	13154	–	43846
Class II	kg FM/yr	120	1462	–	4872
Animal	kg LW	244	2192	–	7070
Biomass[b]	kg FM	30149	29092	0	–
Manure	kg FM	0[c]	0[c]	–	52573
Sales price					
Apple Class I	euro/kg	2.9	2.9	2.9	–
Egg Class I	euro/kg	5.8	5.6	–	5.6
Apple Class II	euro/kg	0.6	0.6	0.6	–
Egg Class II	euro/kg	1.3	1.2	–	1.2
Spent hens	euro/kg	0	0.8	–	0.8
Share of Class I apples	%	88	88	88	–
Share of Class I eggs	%	90	90	–	90
Share of Class II apples	%	12	12	12	–
Share of Class II eggs	%	10	10	–	10
Allocation factor					
Apple Class I (M1)	%	96	88	97	–
Egg Class I (M1)	%	0.9	9	–	96
Spent hens (M1)	%	0	0.2	–	2
Apple Class I (M2)	%	97	97	97	–
Egg Class I (M2)	%	98	95	–	–
Spent hens (M2)	%	0	2	–	–
C-seq					
Compost mix	t C/ha	0.2	1.6	1.2	–
Manure (sheep)[d]	t C/ha	0	0.3	0	–
Manure (bird)[e]	t C/ha	0.01	0.3	0	–
Plant biomass[f]	t C/ha	0.2	0.9	1.2	–
Other biomass[f]	t C/ha	5.4	3.8	3.8	1.6
Pruning	t C/ha	0.8	0	0	–
Leaf litter	t C/ha	2.1	2.1	2.1	–
Roots (fine or coarse)	t C/ha	0.9	0.03	0.03	–
Crop residues	t C/ha	1.6	1.6	1.6	1.6
Outdoor manure deposition	t C/ha	0.03	0.2	0	2.6
Total input to the soil	t C/ha	5.6	5.6	4.3	5.0
C that remains in soil (100-year)	t C/ha	0.2	0.2	0.1	0.03
C-seq (100-year) (soil)	t CO_2-eq/ha	0.6	0.6	0.4	0.1
C that remains in the woody AGB (100-year)	t C/ha	0.10	0.14	0.14	–
C accumulation rate[g]	t C/ha	2.9	3.8	3.8	–
C-seq (100-year) (woody biomass)	t CO_2-eq/ha	0.38	0.52	0.52	0

[a] FM = fresh matter; LW = live weight; p = pieces; kWh = kilowatt-hour; C=Carbon; hrs = hours; ha = hectare.
[b] Refers to the biomass from pruning residues in kg DM exported to the farmers' non-agroforestry farm.
[c] Given the lower stocking densities and larger land area in F1 and F2, it was assumed that all manure excreted by hens inside the bird houses was applied to the agroforestry fields, with no manure exported.
[d] Quantities in kg DM were reported by farmers and then converted into t C/ha as explained in section 2.3.1.4.
[e] Quantities in kg DM were estimated as explained in section 2.3.1.2 and then converted into t C/ha as explained in section 2.3.1.4.
[f] Quantities estimated as explained in section 2.3.1.4.
[g] Over one rotation cycle (15-years).

and cradle-to-retail for the reference year of 2021 and defined as: (i) "1 kg of fresh Class I apples" and "1 kg of fresh Class I eggs". This FU was used because it distinguished the impacts per unit of each product, which enabled direct comparison with other LCAs focused on similar products.

Since the focus of this paper was on the supply side of the value chain (a critical stage for assessing the silvopastoral integration), the system boundary (Fig. 2) was from cradle-to-retail and was divided into two main assessments: food production (cradle-to-farm gate) and post-harvest (cradle-to-retail) both for each product. The post-harvest assessment focused solely on the value chain associated with Class I products. The system boundary considered three resource loops related to the silvopastoral interaction (i.e., manure, land, and feed) and one resource loop mainly related to apple production (i.e., biomass). The inventory data was a combination of farm data reported by farmers and average data from previous years to ensure that the assessment reflected the impacts of standard conditions. Furthermore, activity data for the apple management subphase were solely considered for the high production years (assuming the orchard is 8 years old).

2.3. Life cycle inventory

Activity data (i.e., input quantities) linked to the foreground and background processes for each subphase depicted in Fig. 2 were compiled using a three-step procedure. First, a comprehensive survey protocol was developed and empirical data were systematically collected by the local experts at the two farms. Second, after synthesizing data obtained from the survey protocol, online interviews and questionnaires were conducted with the local experts for further clarification. Third, in-situ farm observations were conducted and complemented by informal interviews with farmers. The interviews were recorded and transcribed for further data extraction and validation. Ongoing dialogue was maintained throughout the assessment with local experts.

2.3.1. Cradle-to-farm gate

2.3.1.1. Modeling approaches for the silvopastoral systems. This paper followed a gross nitrogen (N) mass balance approach (OECD, 2007; Sainju, 2017) and used country-specific methodologies (i.e., Austrian

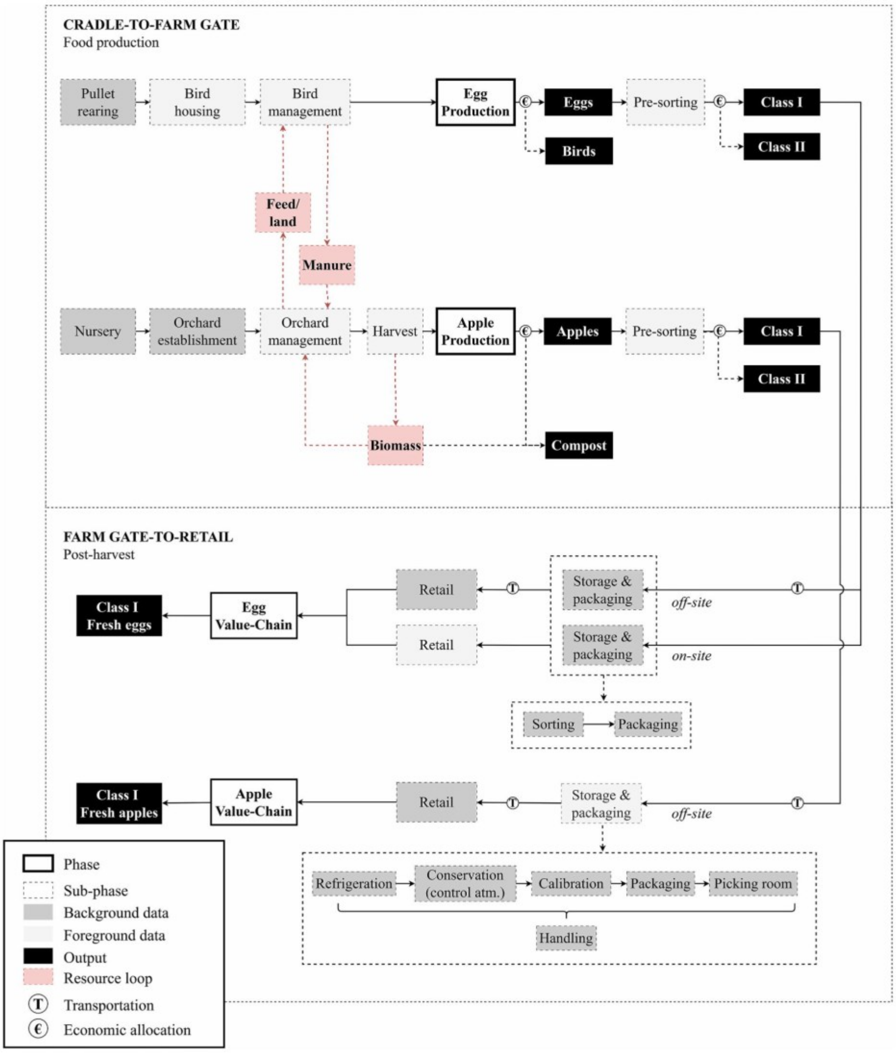

Fig. 2. System boundary. Phases and sub-phases for the apple and egg value chain.

national inventory) and farm-specific data, when available, to calculate N and Phosphorus (P) turnover, emissions, soil C and N changes, and leaching (hereafter referred as flows). Four phases were included (i.e., housing, storage, application, and deposition) to estimate the flows occurring in the production system (at the flock, field, and farm level) and were calculated based on dry matter (DM) content. Input and output data related to the production of apples and eggs at farm level is shown in Table 2.

Two modeling approaches (Fig. 3) were applied at the farm level to distribute the flows associated with apple and egg production across the four phases. In M1, emissions were modeled using the most common modeling approach in the LCA literature. This method used a 'black-box' approach, where the flows were allocated according to the economic value of the products and co-products at the farm gate. In M2, the farm system was partitioned into two subsystems. The emissions associated with the manure excreted during the housing and deposition phase were assigned to the egg subsystem because these emissions were directly associated with the animal production system and were part of the free-range system's normal operations. Whereas emissions associated with the storage and application phase of manure were credited to the system applying/using it (i.e., the apple subsystem). Thus, only emissions from the manure that was stored and transformed were credited to apples, while emissions from manure deposited during grazing were assigned to eggs in M2. This follows the same logic applied to manure excreted during the housing phase, where non-transformed manure was entirely credited to the egg subsystem. Consequently, C-seq and soil N changes

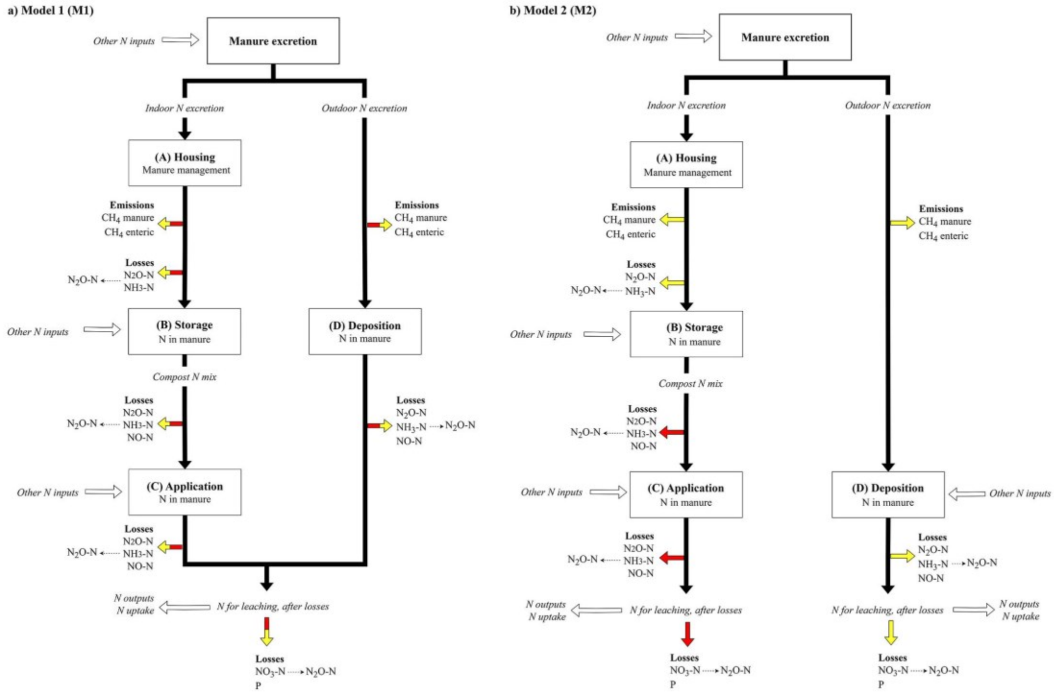

Fig. 3. Farm level models where a) model 1 and b) model 2, including N flows (black arrows) and other N inputs (white arrows). The emissions and losses assigned to the apple- and egg-subsystem throughout the four phases (housing, storage, application, and deposition) are in red and yellow, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

associated with outdoor manure deposition (and crop residues) were also credited entirely to the egg subsystem in M2.

2.3.1.2. N balance at farm level. Table 3 shows the N balances and the corresponding losses of N flows at the farm level. These N flows are then allocated to eggs and apples based on M1 and M2 approaches. The N balance considered the flows of the resource loops specified in Fig. 2. Atmospheric deposition was assumed to be 16 kg N/ha, based on the average values for Eastern Austria (Brentrup et al., 2000), which was aligned with other literature values (Halberg et al., 2010; Knudsen et al., 2010). Moreover, the crude protein (CP) content in apples was assumed to be 2 g per fresh apple, which is consistent with the figures reported in OECD (2019). Additionally, it was assumed that the understory vegetation was a mix of red clover, ryegrass, and herb ley with a biological fixation of 26 kg N/ha (Goh and Ridgen, 1997). This value was corroborated by local experts and other studies (Gentile, 2022). The same N-fixation rate was used across all systems, as the total N input per ha was relatively low in all cases, assuming no negative impact in N-fixation. Data on farm gate prices were used to estimate the allocation factors (AF) for apples (classes I and II) and eggs (classes I and II). Since F1 reported buying spent hens for 2 euros/bird from RS-E, it was assumed that the animals had economic value in both RS-E and F2 for a fair comparison. In F1, the spent hens bought from RS-E were sent to a rendering plant or consumed at the farm at the end of their production cycle. Thus, animals in F1 were considered as residuals (i.e., no emission burden). Economic AFs were used with the farm co-products/residuals (i.e., spent hens) to align with the allocation approach used for class I and II products in all systems and models. In M2, economic allocation followed the same method as in M1, but only included products within each subsystem. In the apple subsystem (F1, F2, RS-A), allocation was between Class I and II apples. In the egg subsystem (F2, RS-E), it was between Class I and II eggs and spent hens (due to their economic value). For the egg subsystem in F1, allocation was only between Class I and II eggs, as spent hens were considered residuals. Equations used for determining the N flows are further described in supplementary data (SD) in Table S1.

Inventory data for the apple and egg production subsystems were compiled for each subphase, as illustrated in Fig. 2. For areas lacking specific information, background data representing upstream generic production values were compiled using the Agribalyse 3 database (Agribalyse, 2020). The annual average animal population was calculated for populations with and without roosters and was determined based on the initial reported batch, the annual replacement of animals at the end of their production cycle, and the annual replacement of animals with pullets or spent hens, following mortality. The calculation of the total N excretion was determined from feed N-intake and N retained by the animals and eggs (excluding roosters), taking into consideration lost eggs. The total annual N excretion during the housing and deposition phase was determined based on the proportion of the year that animals spent indoor and outdoor, as reported by farmers.

The average content of N in live weight (LW) and eggs was assumed to be 28.8 g N/kg LW gain and 18.1 g N/kg egg, respectively (Poulsen et al., 2001). N content in organic feed and forage intake was determined through the CP percentage of DM (since N comprises on average 16% of the total weight of protein) reported in the Danish national feed intake tables for each ingredient. The average DM removal of herbage by laying

Table 3
Estimations of N balances at farm level (within the system boundaries, excluding non-agroforestry land) for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E), including leaching in kg N/ha/year and other N losses.

Input	F1	F2	RS-A	RS-E
	kg N/ha/yr	kg N/ha/yr	kg N/ha/yr	kg N/ha/yr
Concentrates feed	8	77	0	1208
Animal population[a]	1	4	0	42
Atmospheric deposition	16	16	16	16
Compost (other manure)	0	19	0	0
Biological fixation	26	26	26	26
Spelt (litter)	0	0	0	4
Total input	51	143	42	1297
Outputs				
Apples	10	10	10	0
Eggs	2	27	0	309
Animal population	1[b]	6[c]	0	71[c]
Manure (exported)[d]	0	0	0	250
Biomass (exported)[d]	23	20	0	0
Total output	**35**	**63**	**10**	**631**
Balance/Surplus	**15**	**80**	**32**	**666**
Soil N changes[e]	**−16**	**−15**	**−10**	**−3**
Tree growth	**14**	**14**	**14**	**0**
Losses				
N_2O-N (direct)	0.2	1	1	6
NH_3-N	3.8	21	7	214
NO-N	0.9	5	3	29
Total - direct	**5.0**	**27**	**10**	**248**
N_2O-N (indirect)	−0.2	0	0.05	7
Total - indirect	**−0.2**	**0.5**	**0.05**	**7**
Potential leaching NO_3-N	**−20**	**23**	**−2**	**408**[f]

[a] Animal population represents the average N from the annual live weight of animals entering or exiting the farm.
[b] Residual. 50% for human consumption and 50% sent to rendering plant (i.e., economic AF = 0).
[c] Assuming spent hens have an economic value.
[d] Residual. Pruning residues exported to arable land (i.e., economic AF = 0).
[e] Negative values refer to C-seq.
[f] The high leaching is due to RS-E reflecting only the outdoor run area, while F1 and F2 include the total orchard area.

hens during foraging was assumed to be 18 g DM/bird/day based on measurement by Horsted et al. (2006), assuming an average DM content of 12% and a CP of 21%. The total intake was estimated based on the number of days and hours that the animals spent outdoors during the year. It was assumed the same feed and forage intake for roosters, mature hens, and pullets older than 18 weeks of age. Table S2 in SD provides details on feed intake, encompassing the ingredients and their CP. It was assumed that additional calcium was consumed from the soil to fulfill the recommended 7.5 g of calcium intake per bird. This was based on observational field data indicating no signs of calcium deficiency in the eggs. Foraging was assumed to have no impact burdens for the egg subsystem in all models and analyzed systems.

2.3.1.3. Estimating emissions. Emissions during the housing and storage phases were calculated based on the reported N for organic fresh chicken manure (6.1%), green compost (1.2%), and sheep manure (2.7%) (Landwirtschaftskammer, 2010). Depending on the farm, the amount of N in compost – in addition to the N in the collected bird manure excreted indoors – includes N inputs from organic biomass (i.e., pruning biomass) and other manure (i.e., sheep manure). Emission factors (EFs) representing losses associated with N substances are indicated in Table S3 in SD.

NH_3 emissions were calculated using the EMEP/EEA Tier 2 approach, which uses EFs expressed per unit of total ammoniacal nitrogen (TAN) excreted for each of the examined phases (Anderl et al., 2022, Table 219). Direct N_2O emissions from managed soils were calculated using the Tier 2 approach (IPCC, 2019b) and country-specific N excretion rates by considering N in organic fertilizer (i.e., compost) and N in urine and dung deposited during foraging. In addition to direct N_2O emissions from managed soils originating from direct applications, N_2O is also released through two indirect pathways, N volatilization and deposition and N leaching and runoff (IPCC, 2019b). Indirect N losses due to volatilization and leaching from manure management were calculated according to Tier 2 methodology. Tier 1 default EFs for solid manure storage and compost application were used to estimate NO and NO_3 emissions (Anderl et al., 2022). Activity data were then multiplied by a representative EF according to its disaggregated climatic zone (i.e., cool temperate moist climate) and fertilizer type when applicable, as specified in IPCC (2019b). The values were then converted to NH_3, N_2O, NO, and NO_3 emissions (IPCC, 2019a, 2019b). The potential N left for leaching and runoff (NO_3-N) was calculated by subtracting the N outputs, N losses, and N uptake (trees and soil N changes). The soil N changes were assumed to be linked with the below-ground biomass soil C pool (described in section 2.3.1.4), assuming a carbon-to-nitrogen (C:N) ratio of 10. The yearly N uptake in trees (roots, branches, and trunks) was assumed to be 6 g/tree based on Neilsen et al. (2001). These estimates align with figures from Lepp et al. (2024) and Geßl (2020) (i.e., 7.5 g/stone and pome fruit trees).

Total P losses during the application phase at the orchard level were modeled using a P_2O_5 content of 0.3% in the compost (Landwirtschaftskammer, 2010). The EF was estimated using the Tier 1 approach and was defined as the amount of P applied to agricultural fields by foraging and compost and the runoff from soil to water, as explained in PEFCR (2018). For enteric fermentation CH_4, the IPCC Tier 2 approach was used, in which EFs were developed based on Swiss data, as proposed by the Austrian National Inventory (Anderl et al., 2022). This is because Swiss farming practices and conditions were similar to those of Austria. Country-specific EFs were developed for manure management, and the Tier 2 method was used (IPCC, 2019a). Although the excreted manure during housing is managed in multiple systems across the storage and application phases (i.e., pit storage below animal confinement and composting), EFs were calculated according to the fraction of the dominant indoor and outdoor manure management system (i.e., composting with passive windrows and pasture/range/paddock, respectively) and selected according to the dominant climate zone (i.e., temperate moist), as suggested by IPCC (2019a). Default values for CH_4, conversion of N and P emissions, and environmental indicators and characterization factors are in Table S4 in SD.

2.3.1.4. Soil carbon changes and carbon sequestration. A combination of models and methods was used to estimate the potential C-seq and soil C changes. First, the C inputs associated with the belowground biomass (BGB) and woody aboveground biomass (AGB) were quantified. Regarding BGB, the soil C inputs associated with the woody biomass (i.e., leaf litter, pruning residues, and roots) were estimated using allometric functions for apple trees (Ledo et al., 2018). The soil C inputs of the crop residues were estimated according to Mogensen et al. (2018) using coefficients for dry matter allocation of aboveground residues (0.7) and belowground residues (0.4) for perennial grass. The soil C inputs of the compost mix and bird manure were determined based on the N balance and assuming a C:N ratio of 30 and 9, respectively (Brust, 2019; Rynk et al., 2021). Carbon sequestered in AGB, including the trunk, branches, and stem, as well as in BGB (fine and coarse roots) and leaves, was estimated using the Perennial GHG model as described by Ledo et al. (2018) for a complete rotation cycle of 15 years. This model was specifically parameterized for apple trees (Ledo et al. 2018). A key distinction between farms F1 and F2 was the method of harvesting after the rotation cycle. In F1, farmers only removed the woody AGB, and

Table 4
Summary of results for the apple and egg subsystem in each impact category in this paper form cradle-to-farm gate for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E) using Model 1 (M1) and Model 2 (M2).

	CF (kg CO_2-eq)[a]		EP (g PO_4-eq)		AP (g SO_2-eq)		LO	
	per kg apple	per kg egg	per kg apple	per kg egg	per kg apple	per kg egg	per kg apple	per kg egg
This paper:								
Reference	0.104 (0.071)	1.620 (1.613)	0.28	35	1.3	44	0.37	6.8
F1								
M1	**0.094 (0.058)**	**0.192 (0.118)**	*0.35*	**0.7**	**1.0**	**2**	*0.41*	**0.8**
M2	**0.089 (0.051)**	**1.349 (1.324)**	**0.19**	*35*	**0.8**	**36**	0.37	*9.3*
F2								
M1	*0.171 (0.134)*	**0.338 (0.271)**	*1.75*	**3.5**	*2.9*	**6**	*0.62*	**1.2**
M2	*0.109* **(0.068)**	**1.537 (1.520)**	*0.51*	**28**	*1.6*	**33**	0.37	**6.1**

[a] In brackets, net emissions with C inputs contributing to sequestration. Values in cursive or bold are higher or lower relative to the reference system (in regular font), respectively.

after cutting the stem, coarse and fine roots were left in the soil, contributing gradually to C input through decomposition over 100 years. In contrast, in F2, farmers removed both the stem and roots, leaving only fine roots to contribute to soil C. Fine roots have a short lifespan and detach from the main roots annually. It was assumed that fine roots die each year and were replaced by new ones, with dead roots contributing to the soil C pool. Thus, in F2, only fine roots were considered as C input to the soil. The model by Kurz et al. (1996) was used to estimate the proportion of fine roots remaining as the tree aged. Pruning began in year five, with the proportion of biomass pruned from years five to fifteen following the pattern: 5%, 10%, 10%, 10%, 10%, 12%, 12%, 12%, 12%, and 12%. For F2, the pruned biomass was composted, while leaves decomposed directly into the soil, contributing to soil organic C. In F1, the pruned biomass was left on the soil as mulch. C content of dry woody AGB, leaves biomass, and BGB was 0.47%, 0.47%, and 0.44%, respectively (Ledo et al., 2018; Li et al., 2019). C stored in fruits was not included in the analysis, as the fruits were harvested and removed from the field. The C-seq associated with the AGB was estimated using the methodology by Clift and Brandao (2008). Allometric functions used in the study are provided in SD below Table S5.

Second, soil C-seq associated with the C inputs from the BGB was estimated following the methodology proposed by Petersen et al. (2013), where it was assumed that approximately 10% of the C added to soils, under an average air temperature of 8 °C, would be sequestered over 100 years. Given that the mean average air temperature was 10 °C, it was assumed that a 10% estimate was applicable for this paper. To estimate potential C release or sequestration, Danish wheat was chosen as a reference crop like in Mogensen et al. (2014). The difference in total C input from the wheat crop (i.e., 397 kg C/ha/yr) (Mogensen et al., 2018) was calculated for each orchard and reference system and multiplied by 10% to obtain the effect of soil C changes on atmospheric CO_2 over a 100-year perspective. This timeframe was chosen in this paper because it aligns with the 100-year time horizon used for the CF. Data important for the C-seq estimations are specified in Table S5 in SD. A graphical description can be found in Fig. S1 in SD.

2.3.1.5. Land occupation. Land occupation (LO) in m^2a/kg class I product (apple or egg) was estimated for three stages at farm gate: (i) the rearing stage included the LO linked to the background data associated with the concentrates for raising pullets or spent hens, (ii) the husbandry stage included the foreground data associated with the reported concentrates for raising animals during the laying period, and (iii) the farm area included the foreground data associated with the orchard production. Average inventory data from Agribalyse 3 (Agribalyse, 2020) were used for feed production and primary data for the farm area. The total LO was estimated by adding the physical land area used in the three stages. Subsequently, the total LO was allocated to apples and eggs according to their economic value for class I products and then divided by the Class I yield. The AFs were based on those estimated for M1 and M2 (Table 3). For apples and eggs, M1 accounted for the LO in the three stages. For eggs, M2 included the LO from feed production, excluding the farm area. For apples, M2 included the LO from the farm area only,

Table 5
Sensitivity analysis (SA) at farm gate for apples and eggs and percentage differences of methodological choices and assumptions for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

	Apples					Eggs				
		F1		F2			F1		F2	
	RS-A	M1	M2	M1	M2	RS-E	M1	M2	M1	M2
Baseline (kg CO_2-eq/kg)[a]	0.104	0.094	0.089	0.171	0.109	1.62	0.192	1.35	0.34	1.54
a) Manure as waste (M3)	-	-	-1%	-	-4%	-	-	9%	-	6%
Baseline (g PO_4-eq/kg)	0.29	0.35	0.19	1.8	0.5	35	0.7	35	3.5	28
a) Manure as waste (M3)	-	-	-14%	-	-52%	-	-	14%	-	14%
Baseline (g SO_2-eq/kg)	1.33	1.0	0.83	2.9	1.6	44	2.0	36	5.8	33
a) Manure as waste (M3)	-	-	-3%	-	-13%	-	-	13%	-	11%
Baseline (m^2a/kg)	0.37	0.41	0.37	0.62	0.37	6.8	0.83	9.3	1.2	6.1
b) LO AF to eggs	-	-	-1.3%	-	-9%	-	-	10%	-	10%
Baseline (kg CO_2-eq/kg)[b]	0,07	0,058	0,051	0,13	0,07	1,61	0,12	1,32	0,27	1,52
c) Lower AGB C-seq	16%	14%	16%	8%	17%	-	14%	-	7%	-

Red color: SA is higher than the baseline.
Green color: SA is lower than the baseline.
[a]Without C-seq
[b]With C-seq

excluding feed production. A graphical description can be found in Fig. S2 in SD. Furthermore, since a large share of feedstuffs (i.e., maize and wheat) were reported to be produced locally in this paper, it was assumed that there was no direct land use change (dLUC). Similar to Knudsen et al. (2019), the rest of the feed compounds were assumed to be produced at the national level, and soybean compounds were assumed to be imported from China, and thus, dLUC was assumed to be zero.

2.3.2. Cradle-to-retail

Data on post-harvest activities for apples and eggs were only collected for those categorized into class I across two phases: (i) sorting, storage, and packaging and (ii) retail. Distribution-related activities were excluded because a local cooperative was responsible for managing the distribution to retail outlets (as opposed to wholesalers), along with overseeing bulk storage, processing, and packaging operations. Transportation mode, vehicle type, and distance only accounted for off-site activities (i.e., transportation to warehouses in cooperatives and retailers outside farm boundaries). This was because the distances on-site (i.e., at the farm level) were small and considered negligible and accounted for during harvest. A summary of the data used for the two phases is in Table S6 in SD.

During the sorting, storage, and packaging phase, 88% of harvested apples were sorted on-site as class I. All storage occurred off-site at local cooperative facilities. Apples were transported in 300 kg high-density polyethylene pallet bins in refrigerated trucks and stored at 4 °C using controlled atmosphere technology. Data for energy use during storage were based on literature (Boschiero et al., 2019). No packaging materials were used (except for the pallet bins) as class I apples were sold unpacked in bulk. Storage losses were 1%. For eggs, 90% of fresh eggs were sorted on-site as class I. Initial sorting and minimal packaging occurred on-site, with further packaging off-site at cooperative facilities. Packaging materials included corrugated boards, cartons, and labels, transported by various means to the national market. Estimated weights were 390 g for external boxes, 80 g for corrugated boxes, and 15 g for labels (Abbate et al., 2023). Packaging losses were 1%. During the retail phase, class I apples were sold off-site with an assumed 5% loss at retail (Le Féon et al., 2023). Retail considerations included water, energy use, and refrigerated transport. For eggs, 90% of class A eggs were sold off-site, with the remainder sold on-site. Retail considerations included transport to regional and international retailers and water and energy consumption for cooling and cleaning. Retail losses were assumed to be 2% (Kanyama, 2016).

2.4. Life cycle impact assessment

The LCIA was conducted using SimaPro 9.3.0.3 (PRé Sustainability, Amersfoort, Netherlands) and the CML-baseline, which was adapted to the latest IPCC AR6 and Climate Carbon Feedback (CCF) characterization factors. Incorporating CCF entails the consideration of potential additional warming or cooling effects arising from feedback mechanisms. This approach aligns with the overarching objective of capturing long-term climate impacts (e.g., in a 100-year perspective). The CML-baseline LCIA method was chosen because (i) it contains the most common impact categories used in LCA (Merchan and Combelles, 2012), (ii) it has shown strong correlations with other LCIA methods (Pak et al., 2023), and (iii) the combination of its indicators exhibited minimal dependencies (i.e., changes in one indicator did not significantly affect others, making it easier to discern the unique contributions of the selected impact categories) (Pak et al., 2023). Four impact categories were considered: (i) Carbon Footprint (CF) in kg CO_2.eq, (ii) Eutrophication Potential (EP) in g PO_4.eq, (iii) Acidification Potential (AP) in g SO_2.eq, and (iv) Land Occupation (LO) in m^2a. The selection of these impact categories was based on their alignment with the N balance methodology, which enabled a robust assessment of N and P flows and turnover, shedding light on their impacts on the air, soil, and water compartments, as shown in Fig. 1.

The contribution analysis for CF, EP, and AP included nine activities linked to off-farm (OFF) and on-farm (ONF) emissions: (i) feed production (OFF emissions related to the production of feed provided during the husbandry stage, i.e., imported feed), (ii) field operations (ONF emissions related to the operation, production, and maintenance of machineries, including copper sulfate production), (iii) bird rearing (OFF emissions related to the production of pullets or spent hens), (iv) compost mix (ONF emissions related to the storage and application phase according to Fig. 3), (v) bird manure (ONF emissions related to the housing and deposition phase according to Fig. 3, including storage only for RS-E), (vi) buildings (OFF emissions related to the materials used for the hen houses, including litter, and water and electricity consumption), (vii) orchard establishment (ONF emissions related to tree nursery, planting, and orchard establishment), (viii) C-seq in the soil (ONF emissions related to C-seq in the soil), and (ix) C-seq in the woody biomass (ONF emissions related to C-seq in the woody biomass). The contribution analysis for LO included the three stages explained in section 2.3.1.5.

2.5. Sensitivity analysis

A sensitivity analysis (SA) was performed to address uncertainties stemming from key methodological assumptions and data choices. Two variables representing the resource loop were modified individually to analyze their effects on the LCIA: (a) manure classification by modeling it as a waste, hereafter called Model 3 (M3) (Fig. S3 in SD), where all manure-related emissions were assigned 100% to the egg subsystem (assuming no interactions) and (b) LO, where the minimum farm area required for birds were allocated to the egg subsystem in M2 for F1 and F2, in order to determine the land savings from the integration. In addition, (c) only the values for C accumulation rates in the woody AGB were reduced by 64%, from 2.98 to 1.1 t C/ha/yr in F1 and 3.8 to 1.4 t C/ha/yr in F2, aligning more closely with average values reported in some literature (Zanotelli et al., 2015; Yang et al., 2020; McNally and Gentile, 2021). Other parameters (e.g., pruning residues and leaf litter) were the same.

3. Results

3.1. Cradle-to-farm gate

In general, the environmental impacts per kg egg or kg apple in F1 and F2 are lower in most impact categories relative to their reference systems (RS-E and RS-A). The total impact per farm (which is the same across models) is shown in Fig. 4. Results are discussed in more detail under each subsection. A summary of the results from cradle-to-farm gate is shown in Table 4.

The total farm emissions are the same in both models (M1 and M2). However, the way these emissions are distributed to different products (apples, eggs, spent hens) varies significantly, leading to distinct results for each model. In M1, economic allocation distributes the emissions based on the relative market value of the different products, resulting in lower emissions attributed to eggs. In contrast, M2 partitions the farm into two subsystems, where emissions are more directly linked to the processes associated with each product. As a result, emissions per kg egg in M2 are higher compared to per kg egg in M1. The increase in egg-related emissions in F2 compared to F1 was driven by the greater number of hens, feed consumption (0.9 kg N/bird in F2 compared to 0.6 kg N/bird in F1), manure deposition, and pullet rearing.

3.1.1. Climate change

The CF for apples without contributions of C in the soil and the woody biomass is between 0.09 and 0.17 kg CO_2-eq/kg apple (Fig. 5). The contribution analysis showed that field operations contribute between 50 and 91% of the total emissions. The second major contributor

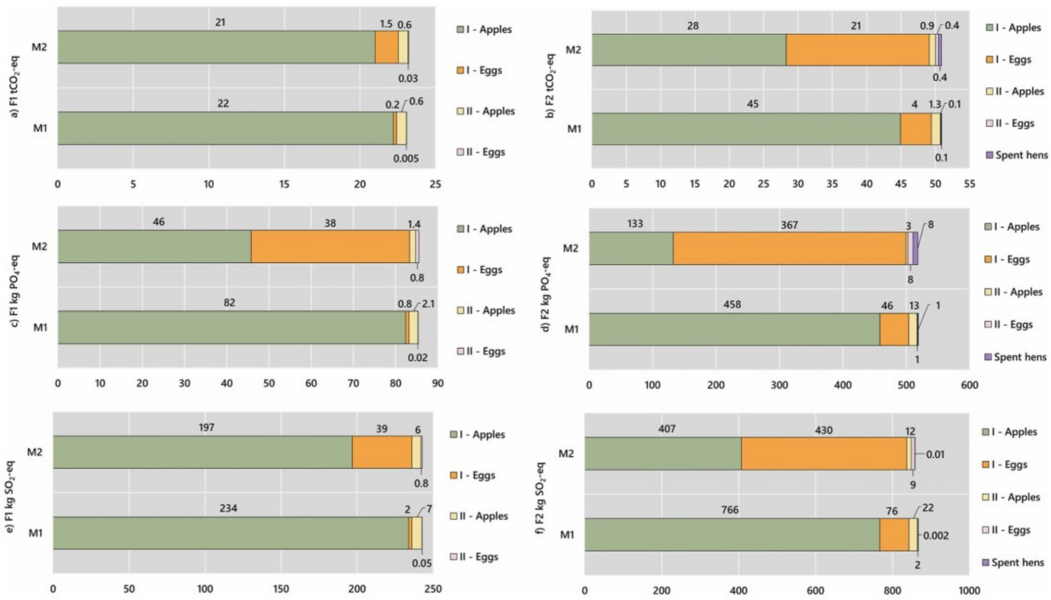

Fig. 4. Total farm emission distribution across class I (I - Apples; I - Eggs), class II (II - Apples; II - Eggs), and other co-products (Spent hens, after the end of production cycle) for Farm 1 (F1) and Farm (F2), including carbon footprint in t CO_2-eq, eutrophication potential in kg PO_4-eq, and acidification potential in kg SO_2-eq. Minor differences between models are attributed to rounding effects during data processing.

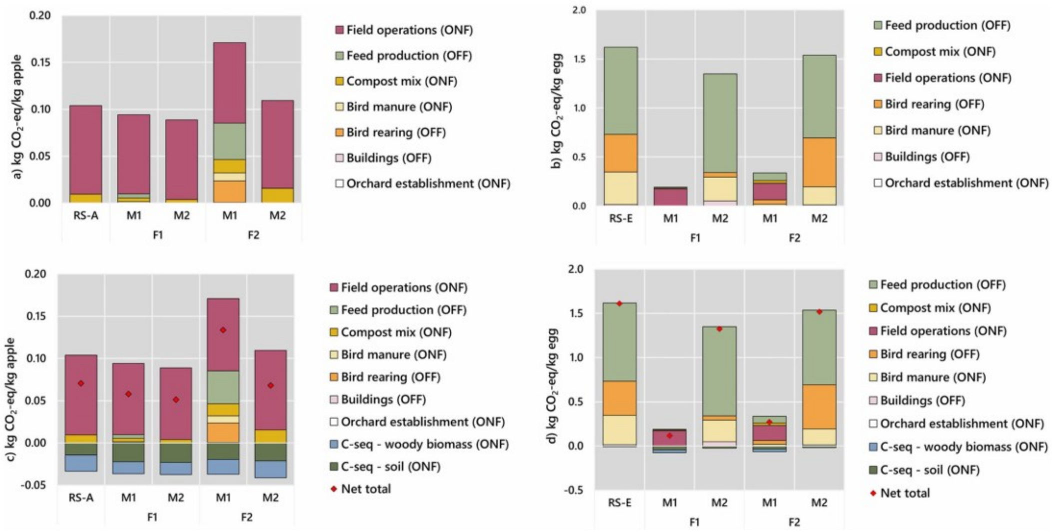

Fig. 5. Carbon footprint (CF) per kg apple (left) and per kg egg (right) in kg CO_2-eq using two modeling approaches (M1, M2) for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E). CF without contribution from C-seq (top) and net CF taking into account contribution from soil C changes and C-seq in the woody biomass (below).

is emissions from the compost mix (4–14%). Due to the economic allocation used in M1, feed production also contributes to the CF of apples (5% for F1 and 23% for F2). Emissions linked to the establishment of apple trees contribute less than 1% of the total emissions in all systems and models. The CF of F1 is lower than RS-A, while F2 is higher than RS-A. Regarding eggs, the CF is between 0.19 and 1.62 kg CO_2-eq/kg egg. Feed production (5–75%), bird manure (1–20%), and bird rearing (0.2–32%) are important contributors across all systems and models. The emissions from field operations are only notable in M1 and minor contributions (less than 4%) are from buildings. Overall, RS-E has a higher CF than F1 and F2. In M1, apples receive a greater share of emissions due to their higher economic value, resulting in lower CF per kg of eggs. F2's higher egg yields further increase the allocation factor, leading to higher emissions per kg apple compared to F1. In M2, differences between F1 and F2 for both apples and eggs are mainly due to management practices, such as compost application and feed production, with F1 benefiting from the use of spent hens and F2 showing higher emissions due to pullets.

Fig. 5 c) and d) show the net CF per kg apple and eggs. C-seq reduces the CF in F1 and F2 by 22–42% for apples and 1–39% for eggs. In the reference systems, C-seq contributes to reductions of 32% and 0.4% for RS-A and RS-E, respectively. Thus, considering all systems and models, emissions were reduced by 22–42% for apples and by 0.4–39% for eggs. C inputs contributing to soil C-seq differ across farms and are shown in Fig. S4 in SD. In F1, most C inputs are from pruning (14%), leaf litter (38%), roots (15%), crop residues (29%), and bird manure (0.7%). In F2, C contributions are from leaf litter (38%), crop residues (29%), and bird manure (9%). In RS-E, 62% of C inputs are from manure and 38% from crop residues. In RS-A, the C contributions are from compost (25%), leaf litter (42%), and crop residues (32%).

3.1.2. Eutrophication potential

EP for apples is between 0.19 and 1.7 g PO_4-eq/kg apple (Fig. 6). Main contributors to EP across all systems and models are compost mix (19–73%), field operations (7–64%), and feed production (25–39%). RS-A has a lower EP, except for F1M2. EP for eggs is between 0.7 and 35 g PO_4-eq/kg egg. Bird manure and feed production represent a considerable share of EP with values between 18-53% and 25–56%, respectively. Bird rearing represents up to 9% of the total impact. Overall, RS-E has a higher EP, with M1 having a lower EP relative to all systems and models. In M1, emissions from egg production are economically allocated to apples where higher egg yields result in a larger allocation factor and share of emissions (0.9%–9% for eggs in F1 and F2, respectively), which also explains the lower impacts per kg egg in F1 and F2. In M2, the differences between F1 and F2 are driven by management choices, such as the fertilization rates for apples and feed production and outdoor manure excretion for eggs.

3.1.3. Acidification potential

AP for apples is between 0.8 and 2.9 g SO_2-eq/kg apple (Fig. 7). Contributions are primarily due to compost mix (22–57%) and field operations (21–74%). Except for F2, RS-A has a higher AP. For eggs, values are between 2 and 36 g SO_2-eq/kg egg, with main contributions from bird manure (11–68%), feed production (5–30%), and bird rearing (1–22%). Like EP, M1 has a lower AP across all systems and models, with RS-E having a higher AP. Similar to EP and CF, in M1, the economic allocation factor places more environmental burden on apples, reducing the AP per kg egg. In F2, the higher egg yields further increase the allocation factor to apples, resulting in higher AP per kg apple compared to F1. Under M2, the direct allocation of impacts to subsystems means that management practices, such as fertilization rates and manure management, drive the differences in AP between F1 and F2 for apples. For eggs, the higher AP in F2 is attributed to the greater amount of manure excreted outdoors and feed production, compared to F1.

3.1.4. Land occupation

Total LO is shown in Fig. 8, with values ranging between 0.3 and 0.6 m^2/kg apple and 0.8–9 m^2/kg egg, respectively. Except for M1, RS-A has the same LO compared to F1 and F2. For eggs, M1 has a lower LO by approximately an order of magnitude than M2 across all systems. The LO for egg production is up to 25-fold greater than that for apple production (except for M1). For apples, feed production during the rearing and production stages of birds contributes approximately 10–42% of the total LO. For eggs, these stages account for about 10–98%. The farm area represents the largest share of LO for apples, ranging from 54 to 100%.

3.2. Cradle-to-retail

Regarding the value chain (Fig. 9), for apples, the CF of the storage and packaging phase accounts for 32–46% of emissions, with an additional 1–3% occurring during retail. For eggs, emissions from storage and packaging range from 11–50%, with retail contributing between 1 and 5%. The storage and packaging phase contributes most of the emissions for EP (2–18% and 0.7–27% for apples and eggs, respectively) and AP (7–20% and 2–29% for apples and eggs, respectively). A summary of the results from farm-gate to retail is shown in Table S7 in SD.

3.3. Sensitivity analysis

Firstly, the SA shows that if manure is classified as a waste (M3), emissions across all impact categories decrease between 1 and 52% for apples and increase between 6 and 14% for eggs (Table 5). Secondly, if the minimum land requirements (in function to the animal density) are allocated to eggs, LO decreases between 1 and 9% for apples and increases 10% for eggs. Third, if the C-seq in the woody AGB is reduced by 64% to align with some literature data (i.e., C accumulation rates around

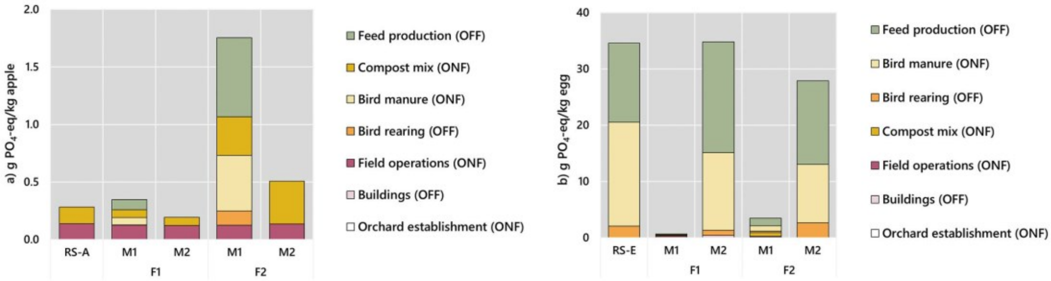

Fig. 6. EP per kg apple (left) and per kg egg (right) in kg PO_4-eq using two modeling approaches (M1, M2) for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

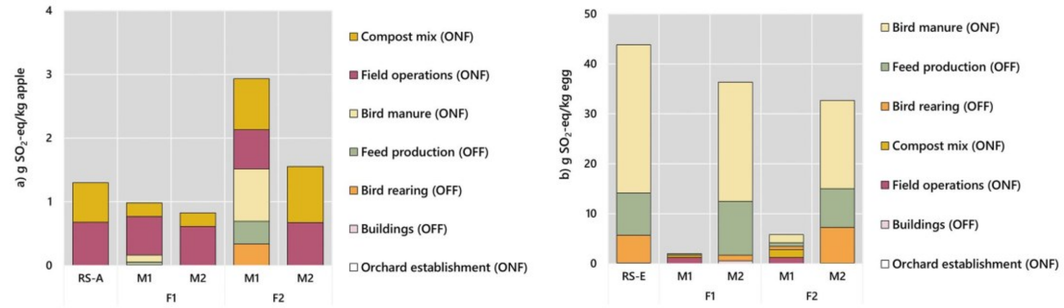

Fig. 7. AP per kg apple (left) and per kg egg (right) in kg SO_2-eq using two modeling approaches (M1, M2) for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

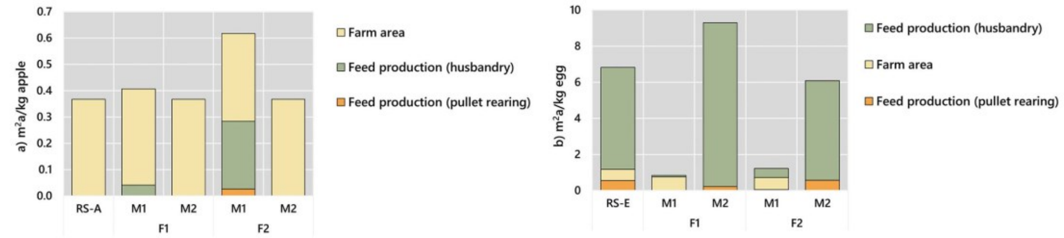

Fig. 8. LO per kg apple (left) and per kg egg (right) in m^2a using two modeling approaches (M1, M2) for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

1 t C/ha/yr), the CF increases for apples and eggs between 8-17% and 7–14%, respectively.

4. Discussion

4.1. *Environmental impacts and comparison with similar studies*

Regarding the first objective, the environmental impacts and CF of two silvopastoral case studies in Austria were assessed, including contribution analysis, post-harvest activities, and C-seq. Below, the results and estimations in this paper were compared with those reported in the organic non-agroforestry literature on apple and egg production (Table S8 in SD). The comparison was conducted from cradle-to-farm gate, as this system boundary was predominantly adopted in the literature.

Regarding the CF, the literature for apple production ranged from 0.07 to 0.15 kg CO_2-eq/kg apple (Alaphilippe et al., 2013; Goossens et al., 2017; Longo et al., 2017) in line with what was found for both the reference systems and the silvopastoral farms in the present paper, whereas Zhu et al. (2018) found a higher CF of 0.87 kg CO_2/kg apple, which could be because of the high fertilization rates and the consequential LCA approach. CF for egg production showed considerable variation in the literature, ranging from 1.30 to 3.42 kg CO_2-eq/kg egg (Dekker et al., 2011; Pelletier, 2017). The CF in kg CO_2/kg egg of 1.62, 1.35, and 1.54 in RS-E, F1, and F2, respectively (when using M2) are within the given range of these results. CF per kg egg in F1 and F2 (0.2 and 0.3 kg CO_2-eq/kg egg respectively) are notably lower for M1 due to the economic allocation. Regarding C-seq, the annual C-seq rate over one rotation period in the woody biomass was between 2.9 and 3.8 t C/ha/yr and C-seq in the soil over 100-year period was 0.03–0.2 t C/ha/yr. The estimates for soil are consistent with values reported in the literature for perennial systems, such as 0.06 t C/ha/yr (McNally and Gentile, 2021). While C in the woody biomass exceeded some literature values, including 0.28–1.5 t C/ha/yr in New Zealand (McNally and Gentile, 2021), 1.37 t C/ha/yr in China (Yang et al., 2020), and 0.58 t C/ha/yr in Italy (Zanotelli et al., 2015), McNally and Gentile (2021) also reported comparable rates in other woody vegetation systems, such as medium-tall hedges (3.5 t C/ha/yr), low hedges (2.2 t C/ha/yr), and untopped trees (3.5–7.9 t C/ha/yr). In addition, the values for the woody biomass in this study align closely with findings from other studies on apple orchards and with similar climate conditions, such as the 3.27 t C/ha/yr observed in the United States (Lakso, 2010) and 4.3 t C/ha/yr in Italy (Scandellari et al., 2016). Several factors may help explain the discrepancies within the literature, such as tree age, tree density, exclusion of root carbon biomass (McNally and Gentile, 2021), and specific management practices, which can significantly influence the extrapolation of results. In general, C-seq remains a topic of significant discussion in the scientific literature, with limited data available for apple systems. Thus a SA on the amount of C-seq in apple trees was conducted, as explained in **section 2.5**.

Regarding EP, EP is less commonly reported in the literature and shows high variations (e.g., 0.03–3.5 and 14–38 g PO_4-eq/kg apple and kg egg, respectively) (Alaphilippe et al., 2013; Longo et al., 2017; Zhu et al., 2018). Results for apples in this paper are within those reported in the literature (except for M1) (see e.g., Leinonen et al., 2012a; Pelletier, 2017; Turner et al., 2022). In contrast, the EP for eggs is slightly above the reported literature values (except for M1), potentially due to lower yields (in this paper, egg yields were around 45% and 11% lower for F1 and F2 respectively, compared with literature values) and the approaches for estimating leaching. These approaches include different characterization factors (e.g., 0.10 kg PO_4-eq in the CML LCIA method or 0.2 kg N-eq in the EF 3.0 LCIA method), assumptions (e.g., between 0 and 20% of excreted N could be assumed as leached) (see e.g., Pelletier, 2017; Brook et al., 2022), or modeling choices (e.g., the inclusion

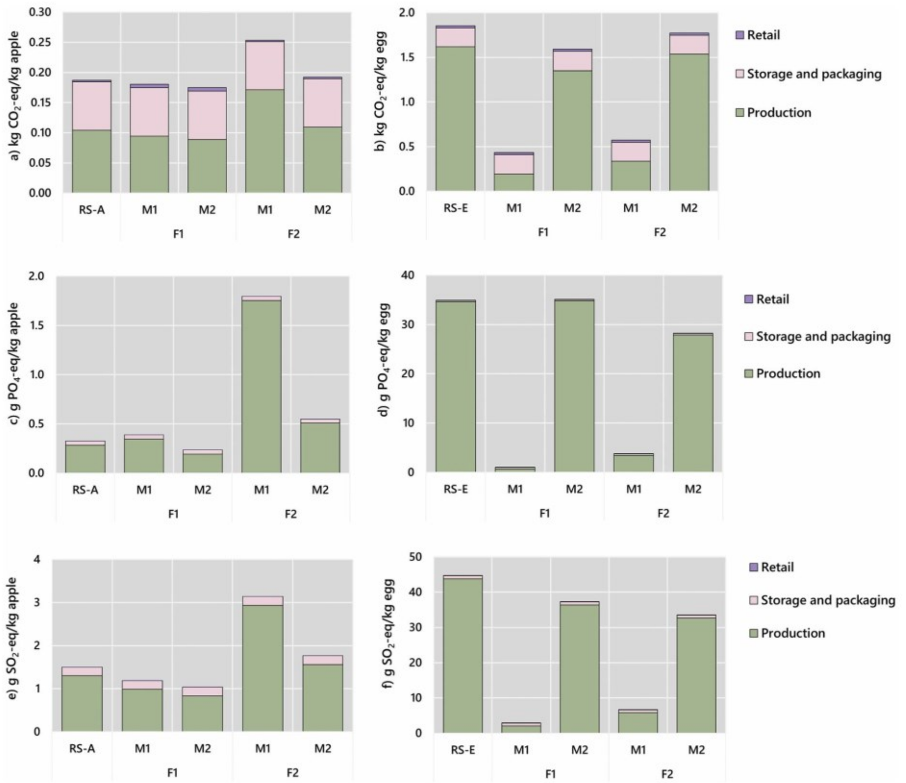

Fig. 9. Post-harvest emissions per kg apple (left) and per kg egg (right) using two modeling approaches (M1, M2) for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E). Production entails emissions from cradle-to-farm gate. Emissions from farm gate-to-retail are linked to storage and packaging and retail.

of climate dynamics or other unaccounted N inputs such as N in irrigation) (see e.g., Brentrup et al., 2000; Kanton St.Gallen, 2013; Serra et al., 2023). In this paper, negative values were estimated for leaching in RS-A and F1 partly due to their substantial C inputs applied into the soil via mulching or composting, using at least 90% of organic plant residues. This influx of C with a high C:N ratio may potentially increase N immobilization for microbial growth, thereby influencing leaching and the EP. Similar to EP, AP is less frequently reported in the literature with ranges between 0.7-1.6 and 42–65 g SO_2-eq per kg apple and egg, respectively. Values in this paper for apples and eggs are lower than in the literature (Dekker et al., 2011; Leinonen et al., 2012a; Goossens et al., 2017; Longo et al., 2017; Pelletier, 2017; Turner et al., 2022). This could be partly due to the lower energy used during field operations for apples and lower NH_3 emissions for eggs.

The system boundary included three resource loops related to the silvopastoral interactions (i.e., manure, land, and feed). One resource loop linked to apple production was also analyzed, namely, biomass. Both biomass and manure served as inputs primarily for fertilizing the orchard or arable lands. The orchards also provided land (outdoor run area) and potential supplemental feed for birds, which influenced the farm-level N estimations. Regarding manure, it was assumed that the excreted manure was evenly distributed in the orchard area. However, hens may not use the full range of the farm area (Bestman, 2017), which could create N hotspots near the houses and around the orchard. Literature values for manure excretion in Austria have been reported as 0.7 kg N/hen/yr (Anderl et al., 2022) which is close to those in this paper (0.5, 0.6, and 0.9 kg N/hen/yr for F1, F2, and RS-E, respectively). Regarding feed intake of hens, the contributions from forage were assumed to be 10%, 8%, and 6% of DM for F1, F2, and RS-E respectively. According to Crawley and Krimpen (2015), around 12–13% of the total DM intake of free-range laying hens can be attributed to forage. Literature values for roughage intake of hens ranged between 0.7 and 72 g DM forage/bird/day (Bosshardt et al., 2022). In this paper, an intake of 6 g forage/hen/day was assumed, however, actual intake can vary due to different factors (e.g., breed and age). The FCRs in this paper were 3.4, 2.2, and 2.8 kg feed/kg egg in F1, F2, and RS-E respectively, which is around the range reported in the literature for organic egg production (only intake of concentrates, i.e. excluding forage intake) (Table S9 in SD). The higher FCRs in F1 may stem from the lower productivity (i.e., 180 eggs per hen) relative to figures cited in the existing organic (276–278 eggs per hen) and conventional (338 eggs per hen) LCA literature (Dekker et al., 2011, 2013; Leinonen et al., 2012b; Abín et al., 2018). Regarding land, although the literature was limited, a study on apple production reports 0.49 m^2a/kg apple (Zhu et al., 2018) and 5–7 m^2a/kg egg (Dekker et al., 2011; Pelletier, 2017), which is in line with what was found for both the reference systems and the analyzed farms.

From a practical perspective, management differences between F1 and F2 have been shown to influence C-seq and nutrient flows. Key strategies for enhancing the soil C pool and optimizing C-seq include minimizing soil disturbance through grafting (as in F1) and recirculating organic inputs, such as pruning residues, through mulching or composting. Nutrient management can be optimized by improving the uniformity of manure distribution across the farm, particularly in the outdoor areas where N hotspots can form near hen houses. Furthermore, optimizing feed management and adjusting concentrate feed quantities according to expected forage intake across different seasons can help reduce GHG emissions and land use. Concerning long-term impacts, while the case studies have the potential to mitigate GHG emissions through C-seq and nutrient recycling, adding more animals to orchards than in current agricultural systems could result in higher overall emissions and greater land use demands due to the additional need for feed. Regarding biodiversity, animals in orchards may serve as biological pest controllers, but they also risk consuming beneficial auxiliary fauna (Bosshardt et al., 2022). Although manure applications can enhance the soil C pool, the carbon input from bird manure into the orchards was relatively small (0.7–9%) and insufficient to offset the substantial negative impacts associated with feed production, among others. In addition, physical disturbances from animals may degrade soil structure over time and could affect water resources through leaching, especially if the system scales with higher animal densities (Sales-Baptista et al., 2016; Bosshardt et al., 2022).

4.2. The effect of different modeling approaches on the silvopastoral systems

Regarding the second objective, two distinct modeling approaches were used for distributing the environmental impact of the entire system between apples and eggs. According to the ISO standards (ISO, 2006a), allocation should be avoided, for example, by creating sub-processes whenever possible. However, when processes cannot be separated, the allocation should be based on the underlying relationships between products, such as economic factors. M1 applied the latter approach, enabling the identification of the main product (with the highest revenue), co-products, and residual products, along with their associated environmental impacts. M1 also took into account resource loops that were interconnected and difficult to separate, such as manure, influenced by food supplementation and land area provided by the orchard, which in turn affected soil and tree growth. However, M1 may lead to assigning a greater share of emissions to high-value products like apples, potentially skewing the assessment in favor of secondary economic activities, such as egg production. This bias can result in overestimating the environmental impacts of apples and underestimating those of eggs. The alternative to economic allocation was to model the multifunctional system by partitioning it into sub-processes (ISO, 2006a). This approach was applied in M2, where emissions were directly linked to the subsystem and the specific product they were associated with. Thus, high-emission activities (e.g., feed production) were not influenced by the economic values or yields of the main product (i.e., apples). However, separating and individually modeling resource loops proved challenging, particularly when manure (associated with the egg subsystem) and plant biomass (associated with the apple subsystem) were mixed in compost, as well as accurately tracking the quantities exported or applied in the orchard.

4.3. Methodological limitations and further research needs

Some methodological limitations were encountered in this assessment and addressed in this section. The methodology for estimating soil C-seq used Danish wheat as the reference crop (for more details read Mogensen et al. 2014). Despite potential geographical differences, wheat is a widely cultivated crop across Europe, including Austria, and has been extensively studied (Heidmann et al., 2001). Furthermore, different pathways for handling woody biomass in apple orchards can influence C release (Le Féon et al., 2023). In this paper, the woody biomass was fully integrated into the compost, thereby reducing uncertainties about the fate of the woody biomass. Other uncertainties regarding C contributions, such as pruning quantities, were minimized by employing well-established models specific to apple orchards (Ledo et al., 2018). Although conventional concentrates were used as proxies in the background database for organic pullet rearing, and French yield-based data was used to estimate LO, the database used in this paper was still the most comprehensive source for European LCAs regarding organic agriculture (Montemayor et al., 2022). Furthermore, although this paper did not model complex compost interactions, including degradation processes, the fertilization rates (i.e., 7, 52, and 47 kg N/ha in F1, F2, and RS-A, respectively) were aligned with standard values for apple cultivation in Denmark (25–40 kg N/ha/yr), France (47–114 kg N/ha/yr) (Alaphilippe et al., 2016), Poland (50 kg N/ha/yr) (Kowalczyk et al., 2022), and Norway (20–65 kg N/ha/yr) (Krogstad et al., 2023).

Regarding future research, it is worth mentioning that the analyzed case studies are not one-size-fits-all. More scenarios, including those without hens, should be explored, such as silvoarable apple production systems (Smith et al., 2014; Pitchers et al., 2017; Staton et al., 2022). In addition, further assessments are needed to evaluate whether the studied systems are 'good enough' from an environmental perspective and operate within their ecosystem's carrying capacity (Hauschild, 2015; Bjørn et al., 2020). Furthermore, other modeling approaches for handling co-products could be further tested and compared, such as system expansion (the manure replaces the organic fertilizer so there are credits from avoided fertilizer production), economic allocation (i.e., assuming manure as a co-product of the system that has value) (Dalgaard and Halberg, 2007; FAO, 2016; Marton et al., 2016; Montemayor et al., 2022), or biophysical allocation. Moreover, while a substitution approach could have been used to account for products displaced by the rendering or on-farm consumption of spent hens in F1, the broader market effects introduced by substitution fall outside the scope of this paper. Also, the consistent economic allocation approach applied across classes and other products ensured a fairer comparison across systems and models. Additionally, given the minimal economic contribution of spent hens (<0.2%), the emissions of the rendering phase were assumed negligible. Moreover, other impact assessment methods for estimating C-seq in the woody biomass and the C opportunity costs (i.e., C-seq if the land had been occupied with native ecosystems instead of pasture or arable crops) (Hayek et al., 2021; Blaustein-Rejto et al., 2023) could be examined. Furthermore, it could be interesting to compare whether integrating trees into an established egg system might be more beneficial rather than the reverse (as in this paper). This approach could involve reducing animal densities and actively replacing the reference system. Alternatively, integrating animals into orchards could be more advantageous during the initial years of orchard development, when trees have higher nutrient demands, rather than during fully productive stages (as were the cases in this paper). Furthermore, the potential benefits of the assessed case studies could be further harnessed if they are developed in harmony with the scientific literature for shifting towards producing less emission-intensive food options while complying with local dietary guidelines (Pierer et al., 2014; Resare Sahlin and Trewern, 2022). This calls for further attention in policy and research papers regarding the prioritization of food quality over quantity and yields.

5. Conclusions

First, this paper assessed the environmental impacts of two silvopastoral case studies (F1 and F2) in Austria, where egg production was integrated into organic apple orchards. The analysis included contributions of carbon sequestration and post-harvest activities. In general, the environmental impacts per kg egg and kg apple in F1 and F2 were lower in most impact categories relative to their reference systems. These impacts were mainly driven by management factors in the production phase of the value chain. Key management factors include the methods of hen sourcing, amount of bought-in concentrates, compost mix and application rates, and between-row management practices. Post-harvest activities accounted for up to 29% of the total impacts for EP and AP, and up to 57% for CC. When accounting for carbon sequestration, emissions were reduced by 22–42% for apples and by 0.4–39% for eggs. Sequestration was predominantly associated with the carbon contributions from plant biomass from apple production (84–99%). Second, two modeling approaches (M1 and M2) at the farm gate were tested and results were compared to standard practices. Overall, F1 and F2 showed different environmental profiles under the same modeling conditions. M1 consistently resulted in higher impacts per kg apple and lower impacts per kg egg relative to M2.

The effects of integrating egg production into apple orchards were also examined through three resource loops (i.e., feed, manure, and land). Although the apple orchards can provide access to forage to animals due to the grassland vegetation, the potential additional feed availability and reductions in bought-in concentrates due to the presence of trees are uncertain, requiring further experiments. While manure had a positive effect on soil, contributing 0.7–9% of the carbon, it was also linked to leaching in M2. Although land was reduced for eggs by 10% due to the land savings from the outdoor run in the farm area, it increased for apples by 4–42% due to feed production during the rearing and production phase in M1.

The environmental assessment in this paper represents specific conditions reported by farmers from two cases and methodological choices made by the authors in this paper. Further case studies are required, and other comparative scenarios and parameters should be further assessed. More development of the LCA modeling and methodological approaches is needed for future assessment of integrated systems.

CRediT authorship contribution statement

Mónica Quevedo-Cascante: Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Teodora Dorca-Preda:** Writing – review & editing, Validation, Supervision, Methodology, Investigation, Formal analysis, Conceptualization. **Lisbeth Mogensen:** Writing – review & editing, Validation, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Werner Zollitsch:** Writing – review & editing, Validation, Investigation, Data curation, Conceptualization. **Muhammad Ahmed Waqas:** Writing – review & editing, Validation, Methodology, Investigation, Formal analysis. **Stefan Hörtenhuber:** Writing – review & editing, Validation, Investigation, Data curation. **Reinhard Geßl:** Validation, Investigation, Data curation, Conceptualization. **Anne Grete Kongsted:** Data curation. **Marie Trydeman Knudsen:** Writing – review & editing, Validation, Supervision, Project administration, Methodology, Investigation, Formal analysis, Conceptualization.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2024.123377.

Data availability

Data will be made available on request.

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B.2. Supplementary materials

Supplementary Material

Life Cycle Assessment and modeling approaches in silvopastoral systems: a case study of egg production integrated in an organic apple orchard

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1. Methodology

1.1. Estimations and databases

Table 1

Equations for N flows.

Annual average animal population (eq. 1)	$$N_T = IB_T * (RPC_T + M_T) \quad (1)$$ Where: $$RPC_T = \frac{Days\ alive}{365}$$ Where: N_T= Average annual population in number; IB_T= Initial number of animals in animal category T in number; RPC_T= Average annual replacement rate after production cycle in animal category T in %; 365= basis for calculating annual RPC_T; M_T= Average annual mortality in animal category T in %

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Total annual N excretion (eq. 2)	$$Nex_{(T)} = (N_{intake\,(T)} - N_{retention(T)}) * 365 \quad (2)$$ Where: $$N_{intake\,(T)} = DMI * \left(\frac{\frac{CP\%}{100}}{6{,}25}\right)$$ $$N_{retention\,(T)} = [\,N_{LW} * WG + (N_{egg} * EGG)]$$ Where: $Nex_{(T)}$= annual N excretion rates in kg N; $N_{intake(T)}$= N intake per head in animal category T in kg; $N_{retention(T)}$= N retained per head in animal category T in kg; 365= Number of days in a year; DMI= dry matter intake per day during a specific growth stage in kg; CP%= Crude protein content in DMI; 6,25= conversion from kg of dietary protein to kg of dietary N; N_{LW}= average content of nitrogen in live weight in kg; WG= average daily weight gain in kg; N_{egg}= average content of N in eggs in kg; EGG= egg mass in kg
Total N leaching at farm level (eq. 3)	$$N_{leaching} = \sum N_{input} - \sum N_{output} - \sum N_{losses} - \sum N_{uptake} \quad (3)$$ Where: $$\sum N_{input} = \sum N_{DMI} + N_{NT} + N_{ATM} + N_{in} + N_{BF}$$ $$\sum N_{output} = N_{out}$$ Where: $N_{leaching}$= net available N for potential leaching in kg N/ha; N_{input}= total sum of N inputs at the farm level; N_{DMI}= Total N from dry matter intake in concentrated feed; N_{NT}= Total N from the average annual population; N_{ATM}= Total N from atmospheric deposition; N_{in}= Total N from imported inputs such as compost or litter; N_{BF}= Total N from biological fixation; N_{output}= Total sum of N outputs at the farm level; N_{out}= N in farm outputs; N_{losses}= total sum of N losses; N_{uptake}= total sum of N uptake for tree growth and soil
NH_3 emissions (eq. 4)	$$NH_3\text{-}N = Nex_{(T)} * EF \quad (4)$$ Where: NH_3-N= NH_3-N losses due to volatilization in kg; $Nex_{(T)}$= Annual N excretion rates in animal category T in kg; EF= Emission factor
Direct N_2O emissions (eq. 5)	$$N_2O_{direct}\text{-}N = N_2O - N_{in} + N_2O\text{-}N_{PRP} \quad (5)$$ Where: N_2O_{direct} – N= annual direct N_2O–N emissions produced from managed soils in kg; N_2O – N_{in}= annual direct N_2O–N emissions produced from N inputs in kg; N_2O – N_{PRP}= annual direct N_2O–N emissions from urine and dung inputs to grazed soils in kg
Indirect N losses due to volatilization (eq. 6)	$$N_2O_{indirect}\text{-}N = N_{volatilization} * EF \quad (6)$$ Where:

	$N_2O_{indirect}$-N= indirect N_2O-N losses due to volatilization in kg; $N_{volatilization}$= amount of N that is lost due to volatilization (NH_3-N) in kg; EF= Emission factor from atmospheric deposition of N on soils and water surfaces
Indirect N losses due to leaching from manure management (eq. 7)	$$N_2O_L\text{-}N = N_{Leaching} * EF \quad (7)$$ Where: N_2O_L-N= indirect N_2O-N losses due to leaching and runoff in kg; $N_{Leaching}$= amount of N that is potentially lost due to leaching (NO_3-N) in kg; EF= Emission factor for leaching and runoff
Total CH_4 emissions (enteric fermentation) (eq. 8)	$$E_T = \sum EF_{(T)} * N_{(T)} \quad (8)$$ Where: $$EF_{(T)} = \frac{GE * \left(\frac{y_m}{100}\right) * 365}{55{,}65}$$ Where: E_T= CH_4 emissions from enteric fermentation in animal category T in kg; $EF_{(T)}$= Emission Factor for the defined animal population T; N= the number of head of livestock species / category T; GE= gross energy intake; Y_m= methane conversion factor; 55.65= energy content of methane
Annual CH4 emission factor for livestock category (eq. 9)	$$EF_{(T)} = (VS_T * 365) * \left[B_{0(T)} * 0{,}67 * \sum_{S,k} \frac{MCF_{S,k}}{100} * AWMS_{(T,S,k)}\right] \quad (9)$$ Where: EF= annual CH_4 emission factor for livestock category T for manure management; VS= daily volatile solid excreted for livestock category T; 365= basis for calculating annual VS production; B_0= maximum methane producing capacity for manure produced by livestock category T; 0,67= conversion factor of m^3 CH_4 to kilograms CH_4; MCF= methane conversion factors for each manure management system S by climate region k; AWMS= fraction of total annual VS for each livestock species/category T that is managed in manure management system S in the country, for productivity system P, when applicable; dimensionless

Table 2

Feed intake, quantified in dry matter (DM), including crude protein (CP) for each ingredient and different mixes for Farm 1 (F1), Farm (F2), and reference system for eggs (RS-E), where: Mix A= summer, Mix B= winter, and Mix C= provided year-round.

	F1	**F2**	**RS-E**	**CP**
Concentrated feed	g DM/bird/day	g DM/bird/day	g DM/bird/day	% of DM
Mix A: Wheat	0	9[a]	0	9
Mix A: Mussel grit	0	1	0	0
Mix A: Oat grains	0	6	0	9
Mix A: Barley	0	8[a]	0	8
Mix A: Soybean	0	18	0	30
Mix A: Maize	0	18[a]	0	8

Mix A: Limestone	0	1	0	0
Mix B: Barley	0	4[a]	0	8
Mix B: Soybean	0	10	0	30
Mix B: Wheat	0	6[a]	0	9
Mix B: Mussel grit	0	0	0	0
Mix B: Maize	0	9[a]	0	8
Mix B: Limestone	0	1	0	0
Mix C: Mussel grit	1	0	0	0
Mix C: Wheat	39[a]	0	25	9
Mix C: Maize	26[a]	0	38	8
Mix C: Corn gluten	0	0	6	20
Mix C: Lucerne cobs	0	0	2	14
Mix C: Pea seed	13	0	2	20
Mix C: Sunflower seeds, unhulled	0	0	7	37
Mix C: Soybean	3	0	21	30
Mix C: Mineral supplements	0	0	11	0
Mix C: Limestone	2	0	2	0
Total	**84**	**90**	**115**	
Forage				
Grass clover, mixed forbs, and chicory	6	6	6	21
Total	**6**	**6**	**6**	
Total feed intake	**90**	**96**	**121**	

[a]Ingredients grown at farm level (excludes emissions from transportation and processing). DM= Dry matter

Table 3

Emission factors for estimating N related emissions across the housing, storage, application, and deposition phases at the flock and field level.

Emission/Phase	Acitivity	Value	Unit	Reference
NH_3-N				
Deposition - excretion	Pasture/Range/Paddock (cattle, poultry, and pigs)	0.25	kg NH_3-N kg^{-1} N	(Anderl *et al.*, 2022)
Housing	Laying hens housing	0.14	kg NH_3-N kg^{-1} N	(Anderl *et al.*, 2022)
Storage - manure	Storage manure	0.06	kg NH_3-N kg^{-1} N	(Anderl *et al.*, 2022)
Application - compost	Compost applied (other organic waste)	0.08	kg NH_3-N kg^{-1} N	(Anderl *et al.*, 2022)
Direct N_2O-N				
Deposition - excretion	Pasture/Range/Paddock (cattle, poultry, and pigs)	0.006	kg N_2O-N kg^{-1} N	(IPCC, 2019b)
Housing	Pit storage below animal confinements	0.002	kg N_2O-N kg^{-1} N	(IPCC, 2019a)
Storage - compost	Composting - Passive windrow (infrequent turning)	0.005	kg N_2O-N kg^{-1} N	(IPCC, 2019a)
Application - compost	N additions from synthetic fertilizers, organic amendments and crop residues, and N mineralized from mineral soil as a result of loss of soil C	0.006	kg N_2O-N kg^{-1} N	(IPCC, 2019b)
Indirect N_2O-N				
Deposition - excretion	N volatilisation and re-deposition	0.01	kg N_2O-N kg^{-1} N	(IPCC, 2019b)

Housing	N volatilisation and re-deposition	0.01	kg N_2O-N kg-[1] N	(IPCC, 2019b)
Storage – compost	N volatilisation and re-deposition	0.01	kg N_2O-N kg-[1] N	(IPCC, 2019b)
Direct NO_2-N				
Deposition - excretion	NO from N applied in fertilizer, manure, and excreta	0.04	kg NO_2-N kg-[1] N	(EMEP/EEA, 2019)
Storage - manure	Stored manure - Laying hens (laying hens and parents) solid manure	0.014	kg NO_2-N kg-[1] N	(EMEP/EEA, 2019)
Application - compost	NO from other organic fertilizers applied to soils (including compost)	0.04	kg NO_2-N kg-[1] N	(EMEP/EEA, 2019)
Indirect N_2O-N from leaching				
Application	Nitrogen leaching (and run-off)	0.011	kg N_2O-N kg-[1] N	(IPCC, 2019b)

Table 4

Default values for CH_4, conversion of N and P emissions, and environmental indicators and characterization factors.

Emission	**Activity**	**Value**	**Unit**	**Reference**
P leaching				
	Phosphorus leaching (and run-off)	0.05	kg P kg^{-1} P	(PEFCR, 2018)
Enteric CH_4				
	Methane Conversion Rate (Y_m)	0.16	%	(Anderl *et al.*, 2022)
	Gross energy intake (GE)	1.80	MJ hen day^{-1}	(Anderl *et al.*, 2022)
	Conversion factor metabolizable energy to gross energy	0.70		(Anderl *et al.*, 2022)
	Energy content of CH_4 (factor)	55.56	MJ kg^{-1} CH_4	(Anderl *et al.*, 2022)
Manure CH_4				
	Volatile Solids (VS) excretion (laying hens)	0.02	kg hen day^{-1}	(Anderl *et al.*, 2022)
	CH_4 producing potential – B_0 (laying hens)[a]	0.39	m² CH_4 kg^{-1} VS	(IPCC, 2019a)
	CH_4 producing potential – B_0 (laying hens)[b]	0.19	m² CH_4 kg^{-1} VS	(IPCC, 2019a)
	MCF pasture/range/paddock	0.47	%	(IPCC, 2019a)
	MCF composting – Passive windrow (Infrequent turning)	1	%	(IPCC, 2019a)
Conversion of emissions	**Substance**	**Value**		**Reference**
	N_2O-N to N_2O	44/28		(Khanali *et al.*, 2020)
	NH_3-N to NH_3	17/14		(Khanali *et al.*, 2020)
	NO_3-N to NO_3	62/14		(Khanali *et al.*, 2020)
	P_2O_5 to P	62/142		(Khanali *et al.*, 2020)
	P to PO_4	3.06		CML-IA baseline
	N to PO_4	0.42		CML-IA baseline
	NO-N to NO	30/14		(Khanali *et al.*, 2020)
	molc H+ to SO_2	0.71		EF 3.0 Method

Environmental indicator	Substance	Value	Unit	Reference
Climate Change	CO_2	1	kg CO_2 eq kg^{-1} FU	IPCC AR6
	CH_4 fossil	29.8		IPCC AR6
	CH_4 non fossil	27.2		IPCC AR6
	N_2O	273		IPCC AR6
Eutrophication Potential	PO_4	1	kg PO_4 eq kg^{-1} FU	CML-IA baseline
	NO	0.2		CML-IA baseline
	NO_3	0.10		CML-IA baseline
	NH_3	0.35		CML-IA baseline
Acidification Potential	SO_2	1	kg SO_2 eq kg^{-1} FU	CML-IA baseline
	NO	0.7		CML-IA baseline
	NH_3	1.6		CML-IA baseline
Land occupation	-	-	m^2a	Agribalyse 3.0

[a]Applied to manure associated with the housing phase.
[b]Applied to manure associated with grazing as recommended by IPCC (2019a) when using pasture/range/paddock MCF.

1.2. Carbon sequestration

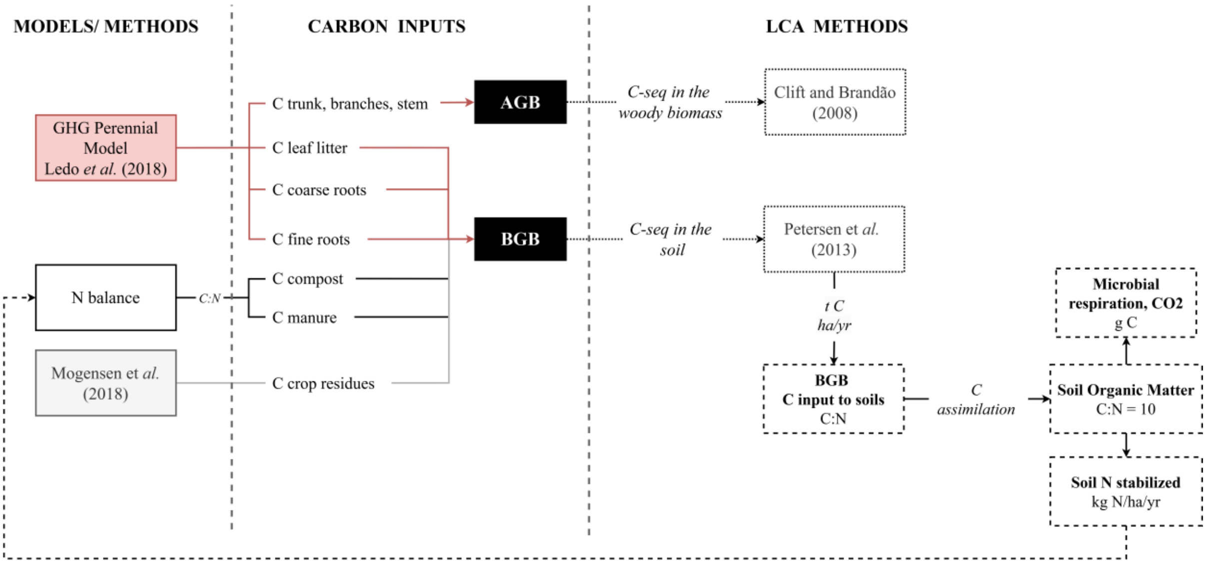

Fig. 1. Models and methods used for estimating carbon inputs associated with the aboveground biomass (AGB) and the belowground biomass (BGB) in order to determine the potential carbon sequestration in the soil and the woody biomass.

Table 5

Characteristics of the apple and egg subsystems important for the estimations of C sequestration for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

Characteristics	Unit	F1	F2	RS-A	RS-E	Source[a]
Land						
Soil (sandy loam)	%	100	0	50	50	PD
Soil (silty loam)	%	0	100	50	50	PD
Soil pH	pH	7	7	7	7	PD
Apple subsystem						
Tree height	m	2	2	2	-	PD
Tree girth	cm	20	20	20	-	PD
Lifespan	yrs	15	15	15	-	PD
Orchard establishment	yrs	Year 1-2	Year 1-2	Year 1-2	-	A
Productive stage	yrs	Year 3-15	Year 3-15	Year 3-15	-	A
Pruning frequency	#/yr	1	1	1	-	PD
Pruning for compost[b]	%	60%	100%	100%	-	PD
Pruning for mulching	%	40%	0%	0%	-	PD
Pruning	Type	Mechanical	Mechanical	Mechanical	-	PD
Removal Year 1-5	%	0	0	0	-	A
Removal Year 6	%	5	5	5	-	A
Removal Year 7-10	%	10	10	10	-	A
Removal Year 11-15	%	12	12	12	-	A
C:N (compost)[c]	ratio	30:1	30:1	30:1	-	A
Compost mix	Type	Mix	Mix	Plant-based	-	PD
Bird manure	%	6	22	0	-	E
Sheep manure	%	0	18	0	-	PD
Pruning biomass	%	94	59	100	-	E
Leave litter[d]	kg/tree/yr	1.9	1.9	1.9	-	E
Biomass from pruning[d]	kg/tree/yr	1.8	1.8	1.8	-	E
Exported	%	80%	42%	0%	-	E
Between-row management	Type	Mulching	Mowing	Mowing	-	PD
Replanting method[e]	Type	Grafting	Uprooting	Uprooting	-	PD
Egg subsystem						
C:N (poultry litter)[f]	ratio	9:1	9:1	-	9:1	A
Forage	days	275	275	-	275	PD
Forage	hr/day	10	10	-	10	PD
Outdoor access	%	67	67	-	67	PD
Indoor access	%	33	33	-	33	PD

[a]PD=Primary data; A=Assumption; E=Estimation.
[b]Not all stored compost was applied. A proportion was exported.
[c]Based on Brust (2019).
[d]In dry matter. Estimated using Ledo *et al.* (2018).
[e]Coarse and fine roots were considered in F1. Only fine roots were considered in F2. Estimated using Ledo *et al.* (2018).
[f]Based on Rynk *et al.* (2021).

Equations for estimating c-seq in the woody biomass are shown below, and are explained in detail in Ledo *et al.* (2018).

Equations for woody above-ground biomass (AGBW) in Ledo *et al.* (2018):

$$AGBW = \left(\alpha_1 age^{\beta_1}\right) * Rw_{AGB} * Rf_{AGB}$$

Where: AGBW represents aboveground woody biomass, and age denotes the age of the aboveground plant parts, measured in years. The parameters α1 and β1 are specific to apple trees (see Table 1). RwAGB and RfAGB account for the impacts of water and nutrient limitations on aboveground biomass. Typically, if there are no water or nutrient limitations and the trees are growing optimally, this value is set to 1. However, in the present case studies, farmers reported nutrient and water limitations during the apple trees' life cycle, so the value was adjusted to 0.7.

Equations for leaf biomass in Ledo *et al.* (2018):

$$Annual\ Leaf\ Biomass_{ev} = \alpha_2 actAGBW^{\beta_2}$$

Where: Annual leaf biomass is the function of AGBW. α2 and β2 are specific parameters for leaf biomass of apple trees. Equations for BGB in Ledo *et al.* (2018):

$$BGB = \left(\alpha_3 age_{root}{}^{\beta_3}\right) * Rw_{BGB} * Rf_{BGB}$$

Where: α3 and β3 are specific parameters for apple trees belowground biomass. The specific parameters are explained in Ledo et al.(2018). RwBGB and RfBGB account for the impacts of water and nutrient limitations on belowground biomass. Typically, if there are no water or nutrient limitations and the trees are growing optimally, this value is set to 1. However, in the present case studies, farmers reported nutrient and water limitations during the apple trees' life cycle, so the value was adjusted to 0.7. BGB refers to the entire root system, including both the coarse roots and the fine roots.

Equations for fine root estimation in Ledo *et al.* (2018):

$$Prop\ fine\ roots_{age_{root}} = 2.73 * age_{root}{}^{-0.841}$$

$$Fine\ root_{root} = Prop\ fine\ roots_{age_{root}} / 100 * BGB_i$$

Fine roots have a short life. It was assumed the fine roots die every year and new fine roots will emerge. Deal will eventually become part of soil carbon.

Equations for pruning:

Pruning (year) = (AGBW – Pruning) year

1.3. Land Occupation

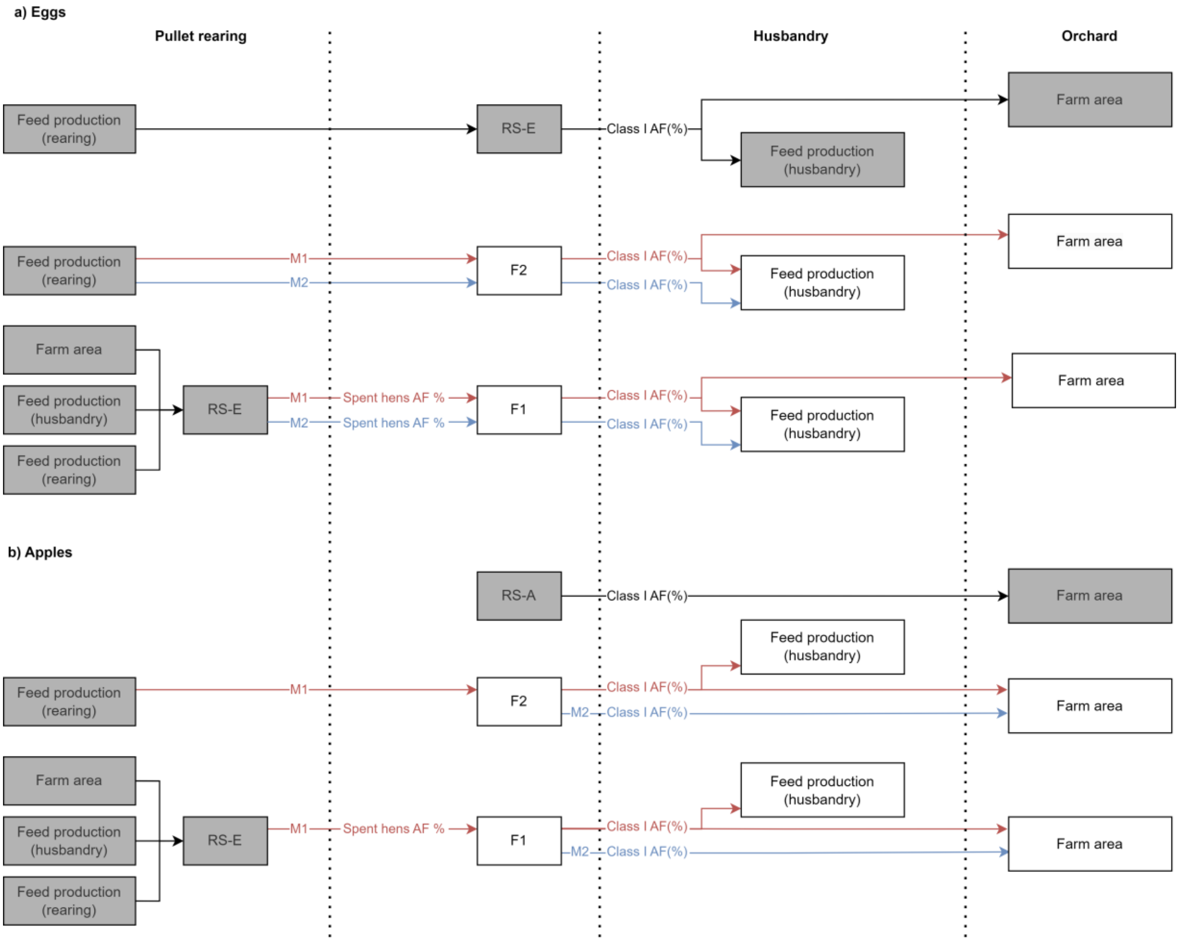

Fig. 2. Background processes (in grey) and foreground processes (in white) associated with land occupation (LO) for class I products during three stages (pullet rearing, husbandry, and orchard production). Red and blue arrows indicates the LO allocated to M1 and M2, respectively. AF = Allocation Factor; F1= Farm 1; F2= Farm 2; RS-E= Reference system for eggs; RS-A= Reference system for apples.

1.4. Post-harvest

Table 6

Input and output data for apples and eggs class I value chain for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

Phase	Activity	Unit	F1	F2	RS-A	RS-E	Source
Input							
Apples							

Storage and packaging	Class I fresh apples	kg/yr	237600	261360	26400	-	PD
Storage and packaging	Transportation (on-site)	tkm	0	0	0	-	PD
Storage and packaging	Transportation (off-site)	tkm	7128	7841	792	-	A
Storage and packaging	Pallet bins	kg/yr	792	871	88	-	EC
Storage and packaging	Electricity	kWh	43790	48169	4866	-	EC
Retail	Transportation (off-site)	tkm	7057	3105	314	-	PD
Retail	Tap water	kg/yr	295477	325025	32831	-	EC
Eggs							
Storage and packaging	Class I fresh eggs	kg/yr	1080	13154	-	43846	PD
Storage and packaging	Transportation (on-site)	km	0	0	-	0	PD
Storage and packaging	Transportation (off-site)	km	25	25	-	25	A
Storage and packaging	Packaging materials	kgkm/kg materials	87	87	-	87	EC
Storage and packaging	Corrugated board box	kg/yr	0	0	-	0	a)
Storage and packaging	Folding boxboard carton	kg/yr	0	0	-	0	a)
Storage and packaging	Label, coated paper	kg/yr	0	0	-	0	a)
Retail	Transportation (on-site)	km	0	0	-	0	A
Retail	Transportation (off-site)	km	153	153	-	153	PD
Retail	Tap water	kg/yr	0,9	0,9	-	0,9	EC
Output							
Apples							
Storage and packaging	Packed and stored off-site	kg apple/yr	235224	258746	26136	-	E
Retail	Stored, packed, and sold off-site	kg apple/yr	223463	245809	24829	-	E
Eggs							
Storage and packaging	Packed and stored off-site[a]	# egg box/yr	1605	18924	-	63081	E
Storage and packaging	Packed and stored on-site[a]	# egg box/yr	176	2079	-	6931	E
Retail	Stored, packed, and sold on-site	kg egg/yr	944	11498	-	38328	E
Retail	Stored, packed, and sold off-site	kg egg/yr	104	1263	-	4211	E
Losses							
Apples							
Storage and packaging		%	1	1	1	-	PD
Retail		%	5	5	5	-	b)
Eggs							
Storage and packaging		%	1	1	-	1	PD
Retail		%	2	2	-	2	b)

[a] 1 egg box contains 10 eggs
a) (Abbate *et al.*, 2023)
b) (Kanyama, 2016)

2. Sensitivity analysis

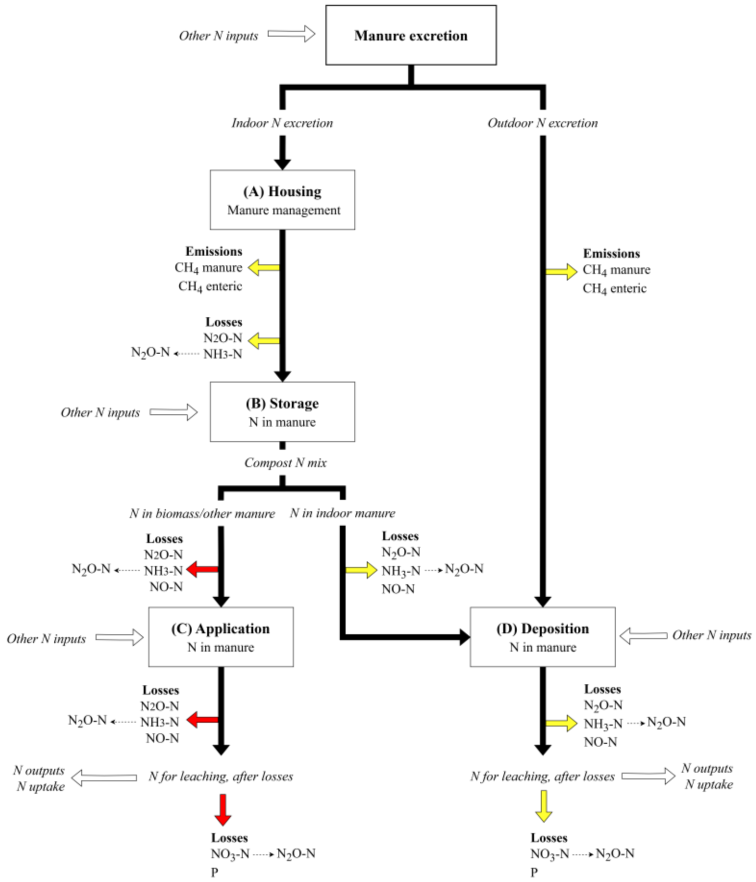

Fig. 3. Alternative farm-gate model 3 (M3), where emissions linked to manure applied and stored are assigned to the egg sub-system.

3. Results

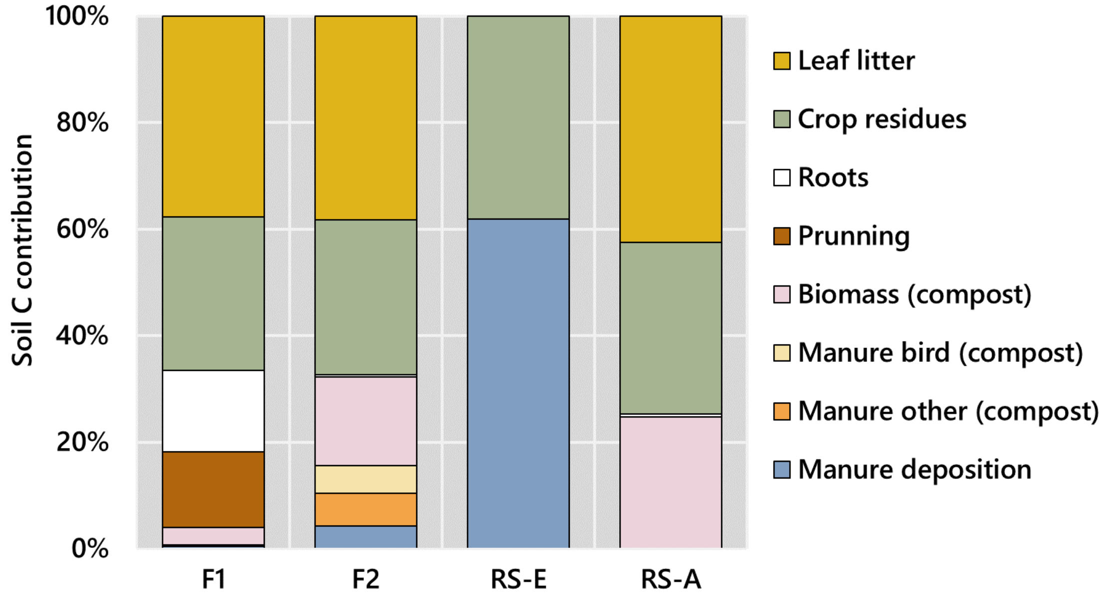

Fig. 4. Share of C inputs contributing to below ground biomass for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E).

Table 7

Summary of results from farm gate-to-retail for the apple and egg subsystem in each impact category in this paper for Farm 1 (F1), Farm (F2), reference system for apples (RS-A), and reference system for eggs (RS-E) using Model 1 (M1) and Model 2 (M2).

	CF ($kg\ CO_2$-eq)		EP ($g\ PO_4$-eq)		AP ($g\ SO_2$-eq)	
	per kg apple	per kg egg	per kg apple	per kg egg	per kg apple	per kg egg
This paper[a]:						
Reference	0.083	0.23	0.04	0.26	0.21	0.87
F1						
M1	0.086	0.24	0.05	0.27	0.22	0.90
M2	0.086	0.24	0.05	0.27	0.22	0.90
F2						
M1	0.083	0.23	0.04	0.26	0.21	0.87
M2	0.083	0.23	0.04	0.26	0.21	0.87

[a]Values in red or green are higher or lower relative to the reference system (in black), respectively.

3.1. Literature data

Table 8

Literature values of LCA studies of non-agroforestry organic apple and egg production systems, from cradle-to-farm gate

	CF ($kg CO_2$-eq)		EP ($g PO_4$-eq)		AP ($g SO_2$-eq)		LO		Yields	
	per kg apple	per kg egg	per kg apple	per kg egg	per kg apple	per kg egg	per kg apple	per kg egg	t/ha	eggs/hen
Literature: Apple										
(Alaphilippe *et al.*, 2013)	0.07	-	1.1[a]	-	-	-	-	-	22	-
(Goossens *et al.*, 2017)	0.15	-	-	-	1.7[b]	-	-	-	33	-
(Longo *et al.*, 2017)	0.10	-	0.03[c]	-	0.7[b]	-	-	-	50	-
(Zhu *et al.*, 2018)	0.87	-	3.5	-	5.3	-	0.49	-	25	-
Egg										
(Dekker *et al.*, 2011)	-	2.54	-	-	-	64	-	7	-	276
(Leinonen *et al.*, 2012)	-	3.42	-	38	-	91	-	-	-	280
(Pelletier, 2017)	-	1.37	-	14	-	47	-	5	-	267-330
(Turner *et al.*, 2022)	-	1.30	-	15	-	47	-	-	-	267-330

[a] Converted from N-eq to PO_4-eq using conversion factor in Table 4.
[b] Converted from molc H+ to SO_2 using conversion factor in Table 4.
[c] Converted from P to PO_4-eq using conversion factor in Table 4.

Table 9

FCR and concentrated feed and pasture intake rates for birds as reported in the literature and data used in this paper for Farm 1 (F1), Farm (F2), and the reference system for eggs (RS-E).

	Concentrated feed intake (g DM/bird/day)	Pasture feed intake (g DM/bird/day)	Production system	Breed	FCR (kg FM concentrated feed/kg FM eggs)
Literature					
(Crawley and Krimpen, 2015)	100-120	-	Organic	-	-
(Gangnat *et al.*, 2020)	115	-	Organic	Lohmann Brown	1.9
(Gangnat *et al.*, 2020)	97	-	Organic	Lohmann Dual	2.0
(Gangnat *et al.*, 2020)	112	-	Organic	Schweizerhuhn	3.5
(Gangnat *et al.*, 2020)	126	-	Organic	Belgian Malines	3.5
(Drinceanu *et al.*, 2016)	117-119	-	Organic	ISA Brown hybrid	3.3-4.3
(Classen, 2017)	87-138	-	Several	Several	-
(Dekker *et al.*, 2011)	-	-	Organic	-	2.6
(Dekker *et al.*, 2013)	101-109	-	Organic	-	-
(Costantini *et al.*, 2020)	130	-	Organic	-	2.5

(Horsted *et al.*, 2006)	-	9-31	Organic	Lohmann Silver	-
(Lorenz *et al.*, 2013)	-	3.6-2.7	-	Bovans and Lohmann Brown	-
This paper					
F1	84	6	Organic	Traditional local	3.4
F2	90	6	Organic	Lohmann Brown	2.2
RS-E	115	6	Organic	Lohmann Brown	2.8

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Appendix C

C.1. Paper III

Development of a Life Cycle Inventory for a Silvopastoral System in the Montado, Portugal

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Keywords: Data; Cork; Cattle, Natural Plantation; Case Study

1. Introduction

The Montado, an agroforestry system (AFS) in Portugal, is known for its ecological complexity and multifunctional land use. Covering around 800.000 hectares, this system integrates woody perennials and trees with pastoral land and, occasionally, small-scale crops (Moreno et al., 2018; Simonson et al., 2018). The Montado is characterized by its sparse tree density and Mediterranean climate, with mild and wet winters and hot and dry summers (Pinto-Correia et al., 2011). It's mainly composed by a silvopastoral landscape, including a range of animals (cattle, sheep, and pigs), alongside woody perennials (cork and shrubs) and other goods (acorns, mushrooms, and honey) typically co-existing in the same unit of land (de Belém Costa Freitas et al., 2020).

Silvopastoral systems in the Montado are valued for their ecological benefits, such as increased biodiversity and reduced fire risk through shrub control (Torres et al., 2017). These configuration can create habitats that support a wide range of plant and animal species (Wilson and Lovell, 2016). While historically these systems were predominantly grazed by sheep, recent shifts influenced by the Common Agricultural Policy have resulted in a greater prevalence of cattle (Sales-Baptista et al., 2016; Arosa et al., 2017; Muñoz-Rojas et al., 2019). This shift has introduced environmental issues due to the high stocking densities (above 0.4 'livestock units' per hectare), including soil compaction and reduced natural regeneration of cork and holm oak trees (Arosa et al., 2017). Additionally, the need for supplemental feed and off-farm fattening activities add further complexity to the environmental profile of the products produced in these systems (Mazzetto et al., 2020).

Life Cycle Assessment (LCA) is a methodology designed to assess the above-mentioned environmental impacts. However, fully capturing the impacts of silvopastoral systems requires detailed and system-specific data, which is currently lacking. LCAs focused on the Montado are scarce, with virtually no studies examining silvopastoral configurations in a natural plantation. Most existing LCA research has concentrated instead on the Dehesa systems in neighboring Spain, primarily evaluating impacts of dairy, meat, or cork production from cradle-to-farm gate (Rives et al., 2011, 2013; Rives, Fernandez-Rodriguez, et al., 2012; Rives, Fernández-Rodríguez, et al., 2012; Escribano et al., 2018, 2022; Horrillo et al., 2020; Reyes-Palomo et al., 2022). The carbon footprint for agroforestry cattle (without carbon sequestration) has been reported to be approximately 17 kg CO2-eq/kg live weight (Eldesouky et al., 2018), 18 kg CO2-eq/kg live weight (Escribano et al., 2022), 10-16 kg CO2-eq/kg live weight (Horrillo et al., 2020)¸ and 9-10 kg CO2-eq/kg live weight (Mazzetto et al., 2020). For the Montado, only five LCAs have been conducted so far, including one on wheat production (Crous-Duran et al., 2019), one consequential LCA of different pasture systems (Morais et al., 2018), and three examining cork production in cork oak woodlands (González-García et al., 2013; Dias et al., 2014; Demertzi et al., 2018). These last studies show that the carbon footprint (without carbon sequestration) can vary between 189-197 kg CO2-eq/t reproduction cork (Dias et al., 2014) and 280-304 kg CO2-eq/t reproduction cork (González-García et al., 2013).

Given that there is a knowledge gap concerning silvopastures in Portugal, the objective of this paper is to establish a foreground Life Cycle Inventory (LCI) for a case study representative to a silvopastoral configuration in a natural regenerated *Montado* system, which focuses on beef cattle and cork production from cradle-to-retail. Developing a representative LCI is an important step in conducting a robust LCA. An incomplete or inaccurate LCI can limit the reflection of complex value-chains, making it challenging to define system boundaries that capture the full range of potential impacts and environmental interactions.

2. Methodology

Data collection involved close collaboration with local network partners in Portugal combined with literature information of context-specific activity data for a *Montado*-based silvopastoral system. Farm data was pre-collected using a survey protocol developed by the MIXED EU-funded project and subsequently validated in this paper through semi-structured interviews and general field observations with local network partners. The study concentrated on two key products, beef and cork, with a specific emphasis on the Alentejo region, which produces 90% of Portugal's cork (Dias et al., 2014). Silvopastures were chosen as the primary production system, as they represent 90% of the Montado landscape, according to the local partners. The focus was on cattle due to a 2.5-fold increase over the past 16 years (Arosa et al., 2017).

For cattle production, the LCI data encompassed both agroforestry-based and non-agroforestry-based farm-gate activities. Within the agroforestry setting, data were collected on cow-calf activities, including details on stocking rates, the animal population structure, and the resources required to sustain the herd. Specific activity data involved the quantity and type of external feed inputs supplied to the cattle (i.e., roughages and concentrates, including their production data), the live weight (LW) of animals sold to fattening farms and slaughterhouses, and the kg of edible product transported to retail (cradle-to-retail).

Regarding cork production, data collection centered on specific aspects of cork oak and holm oak management. Data on tree density was collected to understand the spatial configuration of the silvopastoral landscape. Information on agricultural inputs used for tree maintenance and the composition of pasture grasses was compiled. Data on the type of establishment and stripping timeframes were also collected. Data on cork processing stages were also documented to delineate the cradle-to-retail systems boundary, capturing both the management and extraction phases.

Upstream activity data was also considered, such as fertilizers used during the agroforestry and non-agroforestry phases (e.g., arable land and fattening farms), as well as the type of field operations. Activity data regarding other operations, such as the transport of workers for cork stripping, was also considered. Post-farm gate data (fattening, cork processing, slaughterhouse, and retail) was collected from the literature and validated with local network partners (Horizonte de Projecto, 2017).

3. Preliminary results

The silvopastoral system in the Montado is a conventionally and extensive managed farm with a focus on cork and cattle beef for meat production. Similar to the literature, the central returns come from the cow-calf activity (de Belém Costa Freitas et al., 2020). The system boundary from cradle-to-retail is shown in **Figure 1**. For cork production (**Table 1**), the average yearly yield was estimated to be around a total of 5 t FW/yr or 9.8 kg/ha. These values are considerably lower than those reported in the Montado literature (150-200 kg/ha) partly because of the lower tree density – 5 trees/ha in this paper compared to 50-150 trees/ha in González-García et al. (2013). However, the authors focus on cork woodlands, while Spanish silvopastures report densities between 20-40 trees/ha (Eldesouky et al., 2018; Reyes-Palomo et al., 2022). Stripping starts when the trees are around 30 years old, and the activity repeats every 9 years with an axe. 95% of the proportion of trees are cork oak and 5% holm oak. The trees are established by natural regeneration and are rainfed. Thus, stand establishment operation activities reported in Portuguese systems (e.g., cut-over clearing, furrow- hillocking, planting, fertilizing) (Dias et al., 2014) are not included. Nevertheless, artificial establishments are also common on other farms. Regarding the stand tending stage, no fertilizers are applied to the field and no pruning, cleaning, or thinning operations are conducted. However, other Montado farms can include those operations (Dias et al., 2014). Diesel/petrol production is mainly linked to workers' transport and stripping of the cork. The raw reproduction cork is stored in the agroforestry field area and cut into slabs and then transported to local processors and transformed into cork stoppers.

Table 1. General characteristics, and input and output data for cork production until farm-gate

Parameter	Unit	Value
General characteristics		
Plantation	Type	Natural
Productive tree species	Type	Cork oak
Other tree species	Type	Holm oak
Farm land	ha	681
Grassland (silvopastures)	ha	508

Arable (non-agroforestry)	ha	70
Other (other agroforestry)[a]	ha	103
Total tree density	#/ha	5
Cork oak	#/ha	4.75
Holm oak	#/ha	0.25
Tree height	m	5
Harvest cycle	Years	9
Harvest	#/yr	0.1
INPUT		
Machinery use (cork stripping)	hr/yr	22.2
Workers (traveling to the farm)	hr/yr	25
Fences (maintenance)	hr/yr	26.67
Seeds input	kg/ha	0
Fertilizer N, P, K	kg/ha	0
Limestone	kg/ha	0
Irrigation	m^3/ha	0
Herbicide applications	#/yr	0
Fungicides applications	#/yr	0
Insecticides applications	#/yr	0
Pruning	#/yr	0
OUTPUT		
Raw reproduction cork[b,c]	kg FM/yr	5000

[a]non-productive (buffer zones) and permanent crops
[b]2.3 €/kg (35 €/'arroba', where 1 'arroba'=15 kg)
[c]Total 3000 'arroba' (high-quality cork price)

The cattle system included beef breed cows and beef breed heifers raised for replacement and slaughter, with half being Limousine breed and the other half Angus breed (**Table 2**). The stocking density was around 0.44 animals/ha, slightly lower than the 0.8 animals/ha reported in Spain in Horrillo et al. (2020). Nevertheless, values for silvopastures can range between 0.18-0.74 animals/ha, as reported in other AFS (Escribano et al., 2022). In general, the herd structure can vary significantly from farm to farm, with populations between 12-375 adult cows, as reported in Spanish Dehesas (Reyes-Palomo et al., 2022). Furthermore, calves and cows graze together until weaning and there are no housing facilities, except for some pens for storing roughage and for veterinary activities. Manure and urine are deposited in the soil without any further management. Feed intake is based on grazing and browsing (shrubs) and external concentrates. Concentrates are given ad libitum only for calves (around 4 kg per day) during four months. Cows and bulls generally do not consume any concentrate. However, they may be supplemented during food scarcity periods from August to November. Reproductive animals in Spanish systems have been reported to consume between 136-266.7 kg of fodder/animal and 325.6-357.3 kg of concentrates/animal (Horrillo et al., 2020). Calves spend an average of 6 months in the AFS and are sold at 230 kg live weight (LW) to a specialized fattening farm, which is similar to those values reported in the literature (Horrillo et al., 2020; Reyes-Palomo et al., 2022).

Table 2. General characteristics, and input and output data for beef production until farm-gate

Parameter	Unit	Value
General characteristics		
System	Type	Extensive
Management	Type	Conventional
Breed 1	Type	Limousine
Breed 2	Type	Angus
Calves	# born/cow/yr	0.92
Calving interval	Months	13
Manure management	Type	Left on the field
Stocking density	# animals/ha	0.44
Grazing	# days/yr	375
Grazing	# hr/day	24
Grazing	Type	Continuous
Grass	Type	Grass clover
Legume proportion (grass)	%	50-75
Mortality rate	%	2
INPUT		
Annual beef breed cow	#/yr	200
Annual beef breed bull for breeding	#/yr	3
Annual beef breed heifer for replacement	#/yr	29
Annual beef breed cow	kg LW/yr	600
Concentrate (calf)[a,b]	kg FM/day	4
Concentrate (bull)[b,c]	kg FM/day	7
Concentrates (cow)[b,c]	kg FM/day	4
Forage[d]	# bales/yr	600
OUTPUTs		
Average weight of sold calf[d]	kg LW/yr	230
Average weight of sold discarded cow[e]	kg LW/yr	700

[a]During 4 months period.
[b]Ingredients: Maize, Barley, Sunflower hulls, Wheat bran, Soya beans, Cane molasses, Lucerne, Hydrogenated Fat, Calcium Carbonate, Calcium Phosphate, Sodium Bicarbonate, Sepiolite, Trace elements.
[c]From August to November
[d]Silage produced in 33 ha of arable land (conventional)
[e]600 euros/calf (6 months of age)
[f]800 euros/discarded cow

Cows at the end of their production cycle reached 700 kg LW in the AFS and sold to slaughterhouses. While some farms in *Montado* can have their own fattening facilities, most farms sale their calves to a specialized fattening farm around the region. Calf fattening is based on external feed (no grazing) sourced from concentrates and arable land. Fattened animals were sold to slaughterhouses at 14 months old and 500 kg LW. Manure in fattening facilities is stored and used in the arable land producing roughage and silage for fattened animals. The edible meat products from the local slaughterhouses are sold in domestic and international markets.

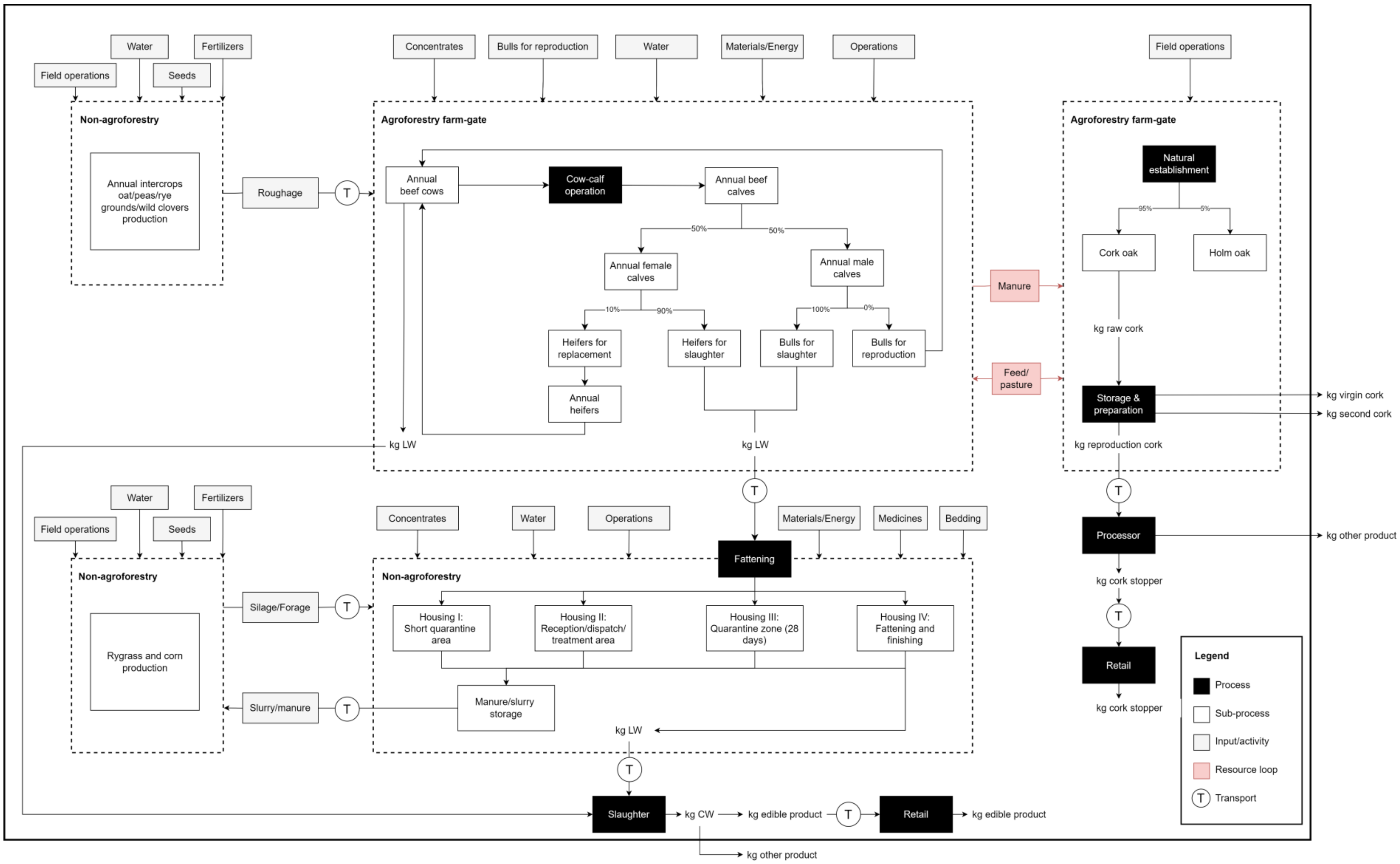

Figure 1. System boundary (in thick black square) of a representative silvopastoral system in the *Montado* from cradle-to-retail for cattle and cork

4. Preliminary discussion

Farms in *Montados* can vary significantly in their focus and land-use practices. While some can prioritize meat production, others may prioritize tourism, hunting, or conservation activities. This variability can introduce modeling challenges for LCAs, as non-agri-food activities can have distinct operational and management approaches. Although this was not the case in the selected farm in this paper, typically, a monetary functional unit and an economic allocation factor would be used to capture the functions of the system and allocate the emissions across different agri-food and non-agri-food economic activities.

Another challenge arises from the mixed-species composition common on some farms. This simultaneous or sequential coexistence complicates LCA modeling, as it requires data specific for each species and because it creates overlapping resource use (e.g., pasture) and nutrient contributions (e.g., manure). For example, if sheep grazed the land the previous year and cattle were grazing it now, the nutrients contributed by sheep manure and any residual effects (e.g., nutrient build-up or depletion) on soil health, plant growth, or carbon levels could influence the current state of the land used by cattle. This residual impact affects the LCA, as the environmental footprint per kg LW should ideally account for these prior management activities. In the present case study, modeling a single-species system (e.g., only cattle on the land) eliminates these complex interspecies interactions which helps in understanding more directly the synergies and trade-offs of cattle in cork oak landscapes. Furthermore, modeling C-seq in the woody biomass is complex due to the centuries-old trees, which have long-established carbon pools. Unlike newly planted systems (e.g., apple orchards), additional carbon accumulation in these systems may be near a stable state. While it is possible to estimate carbon contributions to soil C-seq from manure, the extent to which cattle graze influences C-seq in the woody biomass is uncertain.

Data quality posed another significant limitation in this paper. Farmers are often reluctant to disclose detailed information, particularly regarding feed inputs, such as the quantities and type of concentrate feeds used. To address this issue, secondary data from literature on commonly used feed brands, such as Fonseca, was incorporated. However, this data provides only a general ingredient list without specific percentages, introducing potential inaccuracies. The use of such approximations may lead to under- or overestimations of feed-related impacts, which is a limitation that could affect LCA accuracy. Furthermore, it is also difficult to determine the share of feed linked to pasture or shrub browsing. Thus, sensitivity analysis regarding feed intake should be incorporated. To address this data quality challenge, a potential approach is to estimate the energy requirements for cattle maintenance and production based on values available in scientific literature. By determining the total energy demand of the animals, it is possible to make more accurate analysis about the proportion of energy derived from different feed sources, such as concentrates, pasture, and shrub browsing. For example, if it is determined that concentrates meet 50% of the cattle's daily energy requirements, the remaining 50% could be assumed to come from pasture and shrubs. Thus, when the supply of concentrate is known, a relative good estimate of the theoretical intake of pasture can be made.

Data gaps also emerged in operational activities, particularly for field operations like cork stripping. Information on cork stripping was sourced from neighboring farms (e.g., yearly hours for operational activities) and validated by the local network partners, yet slight variations in practices may still exist between farms, potentially affecting the accuracy of emissions associated with labor and maintenance of cork production. Similarly, data for specialized fattening operations and slaughterhouses were drawn from secondary literature that aligns with the practices reported in Montado systems but may not capture all region-specific distinctions. Other potential limitations include seasonal variability. For example, drought can affect pasture biomass availability, animal productivity, and cork yield, introducing variability that may not be fully captured in a single-season data collection. Thus, data over multiple seasons and years should be further elaborated.

5. Preliminary conclusion

This preliminary LCI addresses the current lack of data specific to silvopastures in *Montado* systems focused on beef cattle and cork production and it is intended as a foundational dataset for subsequent Life Cycle Impact Assessment. This paper highlights important challenges, such as data variability, mixed-species interactions, and allocation issues, that practitioners should consider when modeling these systems.

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In this PhD thesis, the potential environmental and net climate impacts of food from agroforestry systems (AFS) are explored using an attributional Life Cycle Assessment (LCA) framework and a pragmatic mixed-method case study approach. The methodological complexities are addressed by systematically reviewing the literature and testing diverse modeling approaches for handling multifunctionality and estimating the carbon sequestration potential at farm gate. Overall, this thesis provides insights into how the LCA framework captures environmental interactions in AFS from a supply-side and product-level perspective, contributing to a more comprehensive understanding of agroforestry's environmental role in the global food system.