**UNIVERSITA' POLITECNICA DELLE MARCHE**

**Department of Agricultural, Food and Environmental Sciences**

**Ph.D. in Agricultural, Food and Environmental Sciences**

**Specialization:**

**XXXVII Cycle**

# LIFE CYCLE ASSESSMENT OF BIOCONVERSION OF AGRI-FOOD PRODUCTION RESIDUES: EVALUATION AND IMPROVEMENT OF ENERGY AND ENVIRONMENTAL SUSTAINABILITY

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ACADEMIC SESSION 2023/2024

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# DEDICATION

This work
is dedicated with love and gratitude to my dear father,
the late Paul Boakye Yiadom,
and my sweet and loving mother, Victoria Amponaning.

# ACKNOWLEDGEMENTS

This thesis is the culmination of a journey shaped by the effort, support, and encouragement of many incredible individuals and institutions. I extend my deep gratitude to all who played a role in this endeavor.

First and foremost, my deepest thanks go to Prof. Daniele Duca, whose patience, invaluable assistance, and consistent guidance illuminated the path of this project. Your insights and mentorship have been instrumental in shaping this work. I am equally grateful to Dr. Alessio Ilari for your unwavering support, resourcefulness, and precious advice. Your technical expertise, willingness to assist at any hour, and constant encouragement made a difference.

My sincere appreciation also extends to the Department of Agricultural, Food, and Environmental Sciences at Università Politecnica delle Marche, whose resources and facilities were vital for this research. I am especially grateful for their financial support, which made this work possible.

To the LiSEA Group and Biomass Lab, thank you for the opportunity to collaborate with a remarkable team and for providing the essential resources and facilities to bring this project to life. I would like to thank members of my research group, particularly Prof. Ester Foppa Pedretti, Prof. Giuseppe Toscano, Vittorio, Thomas, Elena, Carmine, and Lucia. My gratitude also goes to the Marche Region, whose funding through the PSR Marche Misure 16 for the "BSFlyGreen Project" provided critical support for this PhD.

A special mention to C.O.VAL. M. – O.R.T.O. Verde Scapa for generously providing data, SIGEA for designing and constructing the insect-rearing systems, and Iselqui Technologies for supplying and installing the monitoring sensors. Your contributions were invaluable to this research.

A special thanks to Prof. Valentina Bisinella and her team at the Department of Environmental and Resource Engineering, Technical University of Denmark. Thank you for your warm welcome and collaboration, which made my time in Denmark both enriching and memorable.

On a deeply personal note, I am eternally grateful to my mother, Victoria Ampomaning, whose unwavering prayers and boundless love sustained me through the challenges of this journey. To my late father, Paul Boakye-Yiadom and my late uncle, Clement Johnson Morglo, who passed away during this time, your memories remain a source of strength and inspiration. You are forever in my heart.

To my siblings – Bismark, Peneloppe, Kofi, Godlove, Angel, and Kofi – and my extended family, including my aunt Philomena, my cousins Emefa and Mawuena, and my cherished friends, colleagues, and "super fans" – Fafali, Awurabena, Alex, Helen, Reindorf, Francis, Victoria, Sarah, Evan, Lois, Edith, Erica, and many more – thank you for your motivation, unwavering support, and belief in me and valuable contributions in ways big and small. This work is as much a reflection of your generosity as it is my effort.

I am forever indebted to all who have walked with me on this journey. Thank you for being part of my story.

# DECLARATION

I, Kofi Armah Boakye-Yiadom, declare that the thesis has been composed solely by myself and that the work has not be submitted for any other degree or professional qualification. I confirm that the work is my own and that appropriate credit has been given within this thesis where reference has been made to the work of others. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

# ABSTRACT

This thesis evaluates the environmental performance of frozen horticultural production systems and explores the potential of Black Soldier Fly Larvae (BSFL) bioconversion as an innovative approach to agro-industrial residue valorization. Using the Life Cycle Assessment (LCA) method, the study identifies key environmental hotspots across the horticultural supply chains. Primary data were collected from a representative Italian agricultural consortium specializing in frozen vegetable production.

The findings reveal that cultivation is a major contributor to environmental impacts, particularly in integrated farming systems, due to agrochemical inputs and related emissions. While organic farming showed relatively lower environmental impacts, limitations like small sample sizes necessitate further research to ensure robust comparisons. Processing phases were also identified as hotspots, driven by energy-intensive operations and relative low transformation efficiency. Valorizing these residues can improve system sustainability.

BSFL bioconversion was evaluated as a residue valorization strategy to produce insect biomass. A mass rearing prototype system was designed and tested with the residues and liquid digestate. While results showed promising substrate reduction and larval growth performance, environmental impacts were dominated by electricity for climate control. BSFL protein and fat had higher impacts than conventional protein and fat sources. BSFL also had a higher impact than anaerobic digestion (AD) for residue management. Integrating residual heat from AD emerged as a key strategy to reduce these impacts and make BSFL competitive. The study also identified synergies between BSFL and AD, where BSFL frass serves as a feedstock for biogas production, further promoting circularity.

Despite its current challenges, such as high energy consumption and regulatory barriers, BSFL can be a sustainable protein and fat option. The study underscores the importance of holistic approaches, including improving energy efficiency, adopting precision farming techniques, and integrating BSFL systems to enhance the environmental sustainability of frozen horticultural production chains. This work provides a basis for future studies to refine BSFL technologies and develop scalable and sustainable systems.

# RIASSUNTO

Questa tesi valuta le prestazioni ambientali dei sistemi di produzione orticola di terza gamma ed esplora il potenziale della bioconversione delle Black Soldier Fly Larvae (BSFL) come approccio innovativo alla valorizzazione dei residui agroindustriali. Utilizzando il metodo Life Cycle Assessment (LCA), lo studio identifica i principali punti critici ambientali nelle catene di approvvigionamento ortofrutticolo. I dati primari sono stati raccolti da una società cooperativa italiana rappresentativa e specializzata nella produzione di verdure congelate.

I risultati rivelano che la coltivazione contribuisce in modo determinante agli impatti ambientali, in particolare nei sistemi che seguono i disciplinari di agricoltura integrata, a causa degli input agrochimici e delle relative emissioni. Sebbene l'agricoltura biologica abbia mostrato impatti ambientali relativamente inferiori, limitazioni come le dimensioni ridotte del campione richiedono ulteriori ricerche per garantire confronti robusti. Anche le fasi di lavorazione industriale sono state identificate come punti caldi, i maggiori contributi derivano da operazioni ad alta intensità energetica e di relativa bassa efficienza di trasformazione. La valorizzazione di questi residui può migliorare la sostenibilità del sistema.

La bioconversione BSFL è stata valutata come strategia di valorizzazione dei residui per produrre biomassa di insetti. È stato progettato e testato un prototipo di sistema di allevamento con i residui e il digestato liquido. Mentre i risultati hanno mostrato una promettente riduzione del substrato e prestazioni di crescita delle larve, gli impatti ambientali sono stati dominati dall'elettricità per il controllo del clima. Le proteine e i grassi BSFL hanno avuto impatti maggiori rispetto alle fonti convenzionali di proteine e grassi. La BSFL ha avuto anche un impatto maggiore rispetto alla digestione anaerobica (AD) per la gestione dei residui. L'integrazione del calore residuo dell'AD è emersa come una strategia chiave per ridurre questi impatti e rendere competitiva BSFL. Lo studio ha inoltre identificato sinergie tra BSFL e AD, dove i residui di BSFL fungono da materia prima per la produzione di biogas, promuovendo ulteriormente la circolarità.

Nonostante le sfide attuali, come l'elevato consumo energetico e le barriere normative, il BSFL può rappresentare un'opzione sostenibile per proteine e grassi. Lo studio sottolinea l'importanza degli approcci olistici, tra cui il miglioramento dell'efficienza energetica, l'adozione di tecniche di agricoltura di precisione e l'integrazione dei sistemi BSFL per migliorare la sostenibilità ambientale delle catene di produzione dell'orticoltura congelata. Questo lavoro fornisce una base per studi futuri volti a perfezionare le tecnologie BSFL e sviluppare sistemi scalabili e sostenibili.

# LIST OF PUBLICATIONS

*Journal publications*

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- Ilari, Alessio., **Boakye-Yiadom, K.A**, Duca, D., Foppa Pedretti, E. Valorizing Agro-Industry Residues to Improve the Environmental Sustainability of Frozen Products 2023. In: Ferro, V., Giordano, G., Orlando, S., Vallone, M., Cascone, G., Porto, S.M.C. (eds) *AIIA 2022: Biosystems Engineering Towards the Green Deal*. AIIA 2022. Lecture Notes in Civil Engineering, vol 337. Springer, Cham. https://doi.org/10.1007/978-3-031-30329-6_66

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# LIST OF ABBREVIATIONS

| | |
|---|---|
| ALCA | Attributional Life Cycle Assessment |
| AD | Anaerobic digestion |
| BSFL | Black soldier fly larvae |
| CLCA | Consequential Life Cycle Assessment |
| CRDS | Cavity-ring down spectroscopy |
| EC | Electrochemical |
| ECD | Electron capture detector |
| FID | Flame ionization detector |
| FTIR | Fourier-transform infrared |
| FU | Functional unit |
| GC | Gas chromatography |
| GHG | Greenhouse gas |
| IR | Infrared |
| LCA | Life Cycle Assessment |
| LCI | Life Cycle Inventory |
| LCIA | Life Cycle Impact Assessment |
| MS | Mass spectroscopy |
| NDIR | Non-dispersive infrared |
| NIR | Near-infrared |
| PAS | Photoacoustic spectroscopy |
| PCR | Product Category Rules |
| QCL | Quantum cascade laser |
| SETAC | The Society of Environmental Toxicology and Chemistry |
| SR | Sensitivity ratio |
| TCD | Thermal conductivity detector |
| TDLAS | Tunable diode laser absorption spectroscopy |
| UNEP | United Nations Environmental Program |
| VOC | Volatile organic compound |

Impact assessment categories

| | |
|---|---|
| GWP | Global warming potential |
| CC | Climate change |
| ODP | Ozone depletion |

| | |
|---|---|
| HT_c | Human toxicity, cancer effects |
| HT_nc | Human toxicity, non-cancer effects |
| PM | Particulate matter |
| IR | Ionising radiation |
| POF | Photochemical ozone formation |
| AC | Acidification |
| TEU | Eutrophication, terrestrial |
| FEU | Eutrophication, freshwater |
| MEU | Eutrophication, marine |
| ETF | Ecotoxicity, freshwater |
| LU | Land use |
| WU | Water use |
| FRD | Resource depletion, fossils |
| MRD | Resource depletion, minerals and metals |

# 1 INTRODUCTION

## 1.1. BACKGROUND

Globally, organic waste generation is substantial, with agri-food residues accounting for approximately 5 billion metric tons annually [1]. Managing this vast quantity of residue presents significant economic, environmental, logistical, and health challenges for actors across the food value chain [2, 3]. These residues are often rich in nutrients such as carbohydrates, proteins, lipids, and phytochemicals, which can be recovered or converted into other valuable products like food, feed, bioenergy, and fertilizer. Therefore, valorising agro-food residues has become a crucial strategy for enhancing the sustainability of agri-food production.

Although conventional waste management technologies like landfilling, anaerobic digestion, and composting help in energy and nutrient recovery, they face certain health and environmental limitations that hinder their effectiveness in addressing agri-food residue management. Some of these limitations include a higher requirement for land, greenhouse gas emissions, leachate contamination, and strict process conditions control necessary to process residues (composting and anaerobic digestion) [4, 5]. To mitigate some of these challenges, utilizing black soldier fly larvae (*Hermetia illucens* Linnaeus) for agri-food waste management is gaining popularity as a promising alternative for sustainable organic waste management [6–8]. Therefore, assessing the environmental performance of insect-based waste management can highlight its potential benefits and identify areas for improvement, fostering its broader adoption compared to conventional technologies.

Black soldier fly larvae (BSFL) are a promising solution for valorising agri-industrial waste within a circular bioeconomy framework [9]. Their ability to rapidly convert low-quality organic substrates into high-quality protein and fat-rich biomass and their versatility in feeding on various organic materials make them an attractive option for waste management [10, 11]. Under optimal conditions, BSFL can significantly reduce the volume of organic waste within two weeks, offering economic benefits in terms of logistics and potential applications for residual frass as a soil amendment [12]. Moreover, BSFL can contribute to addressing food security challenges by recovering and upcycling nutrients from organic residues into valuable biomass, which can serve as a source of protein and fat. This makes BSFL a promising strategy for enhancing the sustainability of agri-food production. The BSFL value chain generates several products with

economic potential, including protein sources for animal feed, bioenergy, chitin and chitosan, biopolymers, organic fertilizer, bioplastics, and bioactive compounds [13–15]. These products can serve as sustainable alternatives to conventional resources, promoting a more circular and environmentally sustainable approach to biomass utilization.

The BSFL value chain, while offering potential benefits, is not without its environmental and socioeconomic complexities. Industrial-scale rearing, processing, and valorisation of BSFL for products can be resource and energy-intensive [16]. To promote the sustainable commercialization of BSFL, a comprehensive Life Cycle Assessment (LCA) is essential to accurately assess the environmental impacts of its bioconversion activities and associated post-processes. LCA is a decision-making tool that comprehensively quantifies the environmental impacts associated with each component (input) of the product (output), from raw material extraction to end-of-life disposal [17]. As a valuable tool in promoting sustainability, LCA helps identify areas for improvement and informs decision-making at various stages of the product life cycle [18, 19]. Within the agri-food sector, LCA has gained widespread popularity for assessing the environmental impacts of food products at different stages of their production and consumption [20, 21]. There is a growing interest in applying LCA to evaluate the environmental sustainability of BSFL production and its associated bioconversion activities [16]. However, the sustainability of BSFL production is highly context-specific and influenced by various factors at different stages of the value chain. Previous studies have explored multiple objectives of BSFL production, including biowaste management [22–28], larvae biomass production [29–34], biodiesel production [35], and bioplastic production [36].

The BSFL value chain typically involves upstream processes such as substrate collection and transport, substrate pre-treatment, egg hatchery, and infrastructure. The core processes encompass larval bioconversion activities, harvesting, and processing, including drying, milling, production of frass for soil amendment, and extracting high-value products. Downstream activities include packaging, transportation, product use, and waste management. Moreover, factors like substrate quality, breeding facility conditions, and potential emissions of greenhouse gases and ammonia can impact the environmental sustainability of BSFL bioconversion [16, 37]. Unfortunately, these factors are sometimes excluded in LCA studies on BSFL due to the unavailability of primary data. However, understanding the influence of these parameters is crucial for optimizing the sustainable production of BSFL.

## 1.2. RESEARCH OBJECTIVES

This thesis aims to evaluate the environmental performance of BSFL bioconversion processes applied to various agro-industrial residues. Furthermore, it seeks to provide valuable insights into the environmental implications of utilizing BSFL for sustainable waste management and value-added product generation. The primary goal is to explore the potential for BSFL to enhance circular bioeconomy and sustainability in producing frozen horticultural products. Specifically, the objectives of this doctoral work were to:

- Conduct a comprehensive literature review on LCA studies on BSFL production.
    - Analyse greenhouse gas emissions and LCA studies on BSFL production and activities and identify the main challenges associated with their measurement in BSFL production
    - Identify areas for improvement in applying LCA to BSFL activities for a more holistic understanding of its environmental sustainability.
- Assess the environmental impacts of producing frozen horticultural products using the LCA
- Assess the environmental performance of BSFL production on residues from a horticultural processing industry and identify the key hotspots for improvements using the LCA.
- Evaluate the environmental impacts of producing frozen horticultural products and using BSFL to valorise the agro-industrial residues.
- Assess the potential benefits of incorporating BSFL into food production chains to enhance sustainability.

# 2 LITERATURE REVIEW

The content of this Chapter was published in:

- Sustainability
  *Boakye-Yiadom, K.A., Ilari, A. and Duca, D., 2022. Greenhouse gas emissions and life cycle assessment on the black soldier fly (Hermetia illucens L.). Sustainability, 14(16), p.10456.*
  ***https://doi.org/10.3390/su141610456***

## 2.1. CHAPTER SUMMARY

This Chapter critically reviews greenhouse gas emissions associated with the black soldier fly (*Hermetia illucens* L.) larvae (BSFL) bioconversion activities and the life cycle assessment (LCA) of BSFL production. The review commences with a brief overview of the LCA method. Emphasize is placed on analyzing existing LCA studies on BSFL biotreatment. Methodological variations, common environmental impacts assessed, and identification of key hotspots along the BSFL value chain in these studies are highlighted. Subsequently, the focus shifts to the gases of environmental relevance associated with BSFL biotreatment, investigating how various parameters influence these gases' emissions.

Additionally, the methods and techniques for detecting and quantifying the gases and their respective advantages and disadvantages are summarized. By identifying research gaps and recommending future research directions, this review aimed to contribute to a more comprehensive understanding of the environmental performance of BSFL. Several recommendations are presented, including direct measurement and monitoring of greenhouse gas emissions and harmonizing LCA methodologies for better comparability. Further, given BSFL's potential as a feed substitute, it was suggested that a nutritional LCA be conducted.

However, it is important to note that this review does not include information on horticultural crop cultivation despite its relevance to the broader thesis topic. This decision was made to maintain a focused approach, as the primary innovation of this study revolves around the valorization of agro-industrial residues through BSFL bioconversion. Although horticultural crop cultivation is related to the overall context of improving the sustainability of frozen horticultural products, the emphasis on agro-industrial residues reflects the unique contribution of this study to the field. This narrower focus allows for a deeper examination of BSFL as a potential solution to

sustainable food production, waste management, and the circular economy within the horticultural production chain.

## 2.2. LCA METHOD

### *2.2.1. General principles*

LCA is a structured, internationally standardized methodology that quantifies the environmental burdens of products, processes, systems, or activities throughout their entire life cycles. LCA evaluates environmental impacts, both direct and indirect, from resource extraction through production, use, and end-of-life stages. The ISO 14040/14044 standards [38, 39] provide a regulatory framework for conducting LCA, while the ILCD Handbook offers additional guidance to ensure consistency and quality [40, 41].

### *2.2.2. Framework for LCA method*

LCA is an iterative process with four distinct methodological phases: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and interpretation (Figure 2.1), where the different phases can be repeated until the final objective is met [39]. The phases are not simply followed in a single sequence due to the flexibility of the methodology in allowing for considerable feedback between the phases, making it an iterative process (Figure 2.1).

LCA is useful across various applications, primarily aiming to analyse the interaction between activities or processes and the environment. Its application spans diverse sectors, including agriculture, transport, energy, chemical, nuclear, mining, petroleum, water, electronics, metal, textile, and leather. Some common applications of LCA are summarized in Figure 2.1. LCA enables decision-makers to quantify and evaluate the environmental performance of processes or products, particularly when comparing alternatives. LCA can also be crucial for strategic planning and environmental strategy development by both public and corporate entities. It aids in optimizing, designing, and innovating products and processes, identifying environmental hotspots, and opportunities. Furthermore, LCA assists in selecting relevant environmental performance indicators and measurement techniques and is used in environmental reporting, marketing, and creating frameworks for environmental audits. It also guides policy development to address environmental sustainability challenges and identifies new research and project ideas for sustainability improvements along the supply chain.

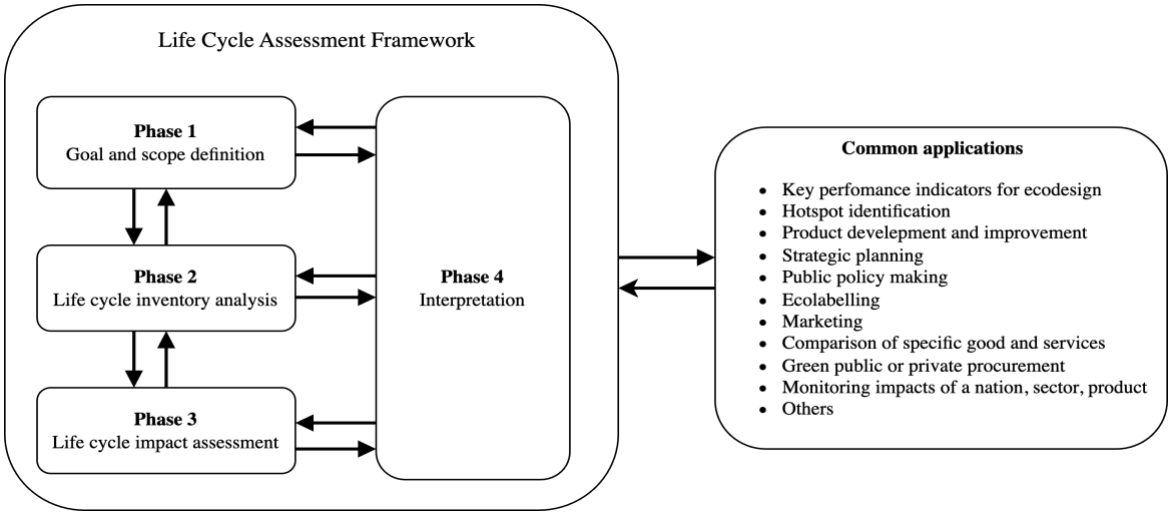



**Figure 2.1.** Structural elements of life cycle assessment.

*2.2.3. Goal and scope definition*

The goal and scope define the essential parameters of the LCA study. The purpose of conducting the study is clearly stated, along with its intended application and target audience [38, 39]. Furthermore, the study's initiator, practitioner, and affected stakeholders are specified. The scope definition establishes the primary characteristics of the intended LCA study, encompassing aspects such as temporal, geographical, and technological coverage, as well as the employed mode of analysis [42, 43]. The study's scope must include a statement regarding the system's function with its corresponding functional unit, the reference unit upon which emissions and extractions will be based. The system boundaries, which define what will be included or excluded from the study, are specified. Additionally, the definition of goal and scope must specify the principles for handling multifunctionality, data quality requirements, and the selection of impact categories and methods for impact assessment [42, 43].

*2.2.4 Attributional versus consequential approaches*

Generally, there are two types of LCAs with different modelling approaches: Attributional LCA (ALCA) and consequential LCA (CLCA). The two alternative modelling perspectives have been developed and extensively discussed in LCA.

ALCA also known as the accounting or descriptive approach focuses on quantifying the environmental impacts associated with a specific product or process, based on its current state. It provides a static view of the system's environmental performance, making it suitable for comparing different products or services. ALCA is mostly used in a retrospective context to evaluate existing

systems [44, 45]. CLCA also known as the change-oriented modelling approach, on the other hand, considers the potential environmental impacts of future changes to a system. It assesses the environmental consequences of specific decisions or interventions, such as the introduction of a new technology or policy. CLCA requires a deeper understanding of market dynamics, technological advancements, and the broader system context. CLCA is more suitable for assessing the environmental implications of future scenarios and policy decisions [43–46].

The choice of modelling approach, either attributional or consequential LCA, significantly influences the results and their interpretation. While these two approaches are distinct, a comprehensive understanding of a product's or process's environmental impact can be gained by considering both perspectives.

*2.2.5 Handling multifunctional systems*

When conducting attributional LCA, a common challenge arises when a process generates multiple products (multifunctionality). In such cases, allocating environmental burdens to individual products becomes necessary. To address this, ISO 14044:2006 [38] and the ILCD guidelines [40] recommend a hierarchical approach:

- Avoiding allocation through subdivision: This involves dividing the process generating co-products into sub-activities that deliver unique co-products, allowing for discrete analysis of inputs and outputs for each sub-activity.
- System expansion: This approach broadens the analysis to include separate, parallel processes that yield functionally equivalent products to the co-products, allowing impacts of the co-produced product to be subtracted from the main product. Note that in CLCA, system expansion is used by default, thereby inherently avoiding allocation.
- Allocation based on physical relationships: This method uses quantitative relationships that reflect how inputs and outputs vary with changes in the production of each product in the system.
- Allocation based on other relationships: When physical relationships are not applicable, allocation may be based on alternative factors, such as economic value.

It is important to note that system expansion is a default approach in CLCA, as it focuses on the incremental environmental impacts of changes in production or consumption. Thus, allocation is not performed in CLA. Moreover, ALCA generally involves modelling the background system with average processes, representing an average supply chain. In contrast, CLCA models the

background system using marginal processes, which reflect the processes that would be added or removed in response to a change in demand for a product [43, 46].

*2.2.6. Life Cycle Inventory (LCI) Analysis*

The life cycle inventory (LCI) analysis process involves the quantitative description of the flow of matter, energy, and pollutants related to the processes in the system under study [38, 39]. These inputs and outputs directly relate to the defined functional unit and any requirements related to the goal and scope of the research. LCI analysis involves collecting data for the individual unit processes and the calculation procedures to normalize the data. The result of this phase is a table that quantifies the relevant inputs and outputs of the product system normalized to the selected functional unit [46, 47].

*2.2.7. Life Cycle Impact Assessment (LCIA)*

Life cycle impact assessment (LCIA) translates the life cycle inventory analysis results into environmental impacts, such as climate change, acidification, and eutrophication. It evaluates the magnitude of the contribution of each elementary flow to a potential impact on the environment. This process involves selecting the impact categories, classification of the emissions, substances, and resources that affect each selected impact category as well as characterization, normalization, weighting, and grouping. Modelling results are calculated during characterization, with an optional normalization indicating the share of the results in a global or regional context. Impact category indicator results can be also grouped and weighted to reflect societal preferences across various impact categories. Impacts can be categorized as either midpoint or endpoint. Midpoint indicators focus on the potential harm to a single environmental issue. In contrast, endpoint indicators show the potential impact of aggregated midpoint indicators on broader environmental issues [46, 47].

*2.2.8. Interpretation*

The interpretation phase involves the interpretation of the results obtained and evaluating uncertainties to improve the assessment. The LCIA results are elaborated and interpreted in line with the goal of the study, main assumptions, and modelling choices made. Interpretation involves completeness check, consistency check, sensitivity check, identification of significant issues, and conclusions, limitations, and recommendations. The key parameters and improvement options can be identified through hotspot analysis, contribution analysis, and dominance analysis. Uncertainty analysis is conducted to study the influence of the individual uncertainties of the input parameters on the results. Sensitivity analysis, including scenario analysis is done to evaluate the influence of

the chosen system boundaries, assumptions, and modelling choices like allocation key. Sensitivity and uncertainty analyses are essential parts of the LCA interpretation phase. Sensitivity analysis helps assess how input uncertainties impact outcomes, while uncertainty analysis establishes the confidence range for results. Both can be applied in the LCI and LCIA phases to identify potential sources of variation, evaluate data quality, and gauge the robustness of the study [46, 48]. However, no standardized techniques for these analyses currently exist.

*2.2.9. Some limitations of LCA*

LCA, while a valuable tool, is not without its limitations. The inherent complexity of real-world systems often necessitates simplifying assumptions and scenarios, which can introduce potential biases. Variations in methodological choices, such as the selection of functional units, system boundaries, allocation methods, and impact assessment categories can also influence results and may lead to misinterpretations, particularly among non-experts. These decisions are often based on the researcher's discretion and may lead to additional biases. Moreover, LCA studies are resource-intensive, requiring significant data collection and analysis. Insufficient or poor-quality data can compromise the reliability of conclusions [49].

LCA studies tend to be case-specific. Thus, LCA results may not represent the best-case scenario for a particular product, as all studies may not consider novel production systems with the potential for significant environmental improvements. In addition, communicating LCA results can also be challenging, especially regarding multiple environmental impacts, and may require careful consideration and explanation to avoid misinterpretations. Sometimes, comparing products based on LCA results can be challenging due to trade-offs between different environmental impacts. Identifying which impacts should be prioritized also requires careful consideration and may vary depending on specific contexts. LCA is a good but incomplete tool for measuring sustainability. It may not provide definitive limits for acceptable environmental impacts at a societal level [50, 51].

Addressing these limitations requires a nuanced understanding of LCA's strengths and weaknesses and ongoing efforts to improve methodologies and data quality. By acknowledging these challenges, researchers can enhance the credibility and applicability of LCA studies. Several advancements are underway to harmonize LCA methodologies to improve the application of LCA in assessing the environmental performance of products and services. For example, the United Nations Environment Program (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC) have an ongoing joint initiative, "Life-Cycle Initiative," with the aim of easing

access to inventory data and providing standardized impact assessment procedures for practitioners by the United Nations Environmental Program (UNEP) and SETAC [52].

## 2.3. LCA STUDIES ON BSFL

Over the past few years, several LCA studies have been conducted to evaluate the environmental profile of insect-based products, including BSFL. A BSFL value chain can generate several high-value products through additional processing. Crude protein from the insect cake can be further processed to obtain bioplastic film, protein hydrolysate, and emulsifying and foaming agents [36, 53, 54]. Chitin from insect meals can also be refined to produce an edible film, surgical thread, binders, and chitosan [11]. Crude fats can also be refined into biodiesel and lubricating agents [10, 55, 56]. However, LCA has only been applied to a few BSFL products; Table 2.1 summarises LCA studies related to BSFL. The studies were conducted within the last decade, mainly in Europe. The use of BSFL as animal feed, as well as its derived products, is limited by the type of feeding material for its rearing. EC No. 1069/2009 restricts the use of food waste as feeding material, especially when it contains or is derived from catering waste [57]. However, agri-food residues are permissible. Thus, half of the studies analysed focused solely on the environmental impact of organic waste bioconversion by the BSFL for waste treatment due to the substrate typology, though insect products could also be obtained.

### *2.3.1 LCA methodological choices*

This section highlights some key methodological aspects that can significantly influence the results and interpretations of LCA studies on BSFL bioconversion.

#### 2.3.1.1 Goal and scope definition

The goals of the LCA studies analysed on BSFL can be broadly categorized into three groups: studies focused on waste management, those centred on conventional BSFL biomass production and its related products, and studies addressing both areas. Of the studies reviewed, 43% focused on biowaste treatment, another 43% on BSFL production and related products, and 14% covered both. Understanding the specific goals of each study is essential for scoping the LCA and accurate interpretation of results, and therefore, it should be communicated clearly.

#### 2.3.1.2 Geographical and temporal scope

The geographical and temporal scope of an LCA study significantly influences data quality and the accuracy of results. The foreground and background systems should be aligned in terms of spatial

and temporal boundaries to ensure consistency [43, 46]. Most of the analysed LCA studies on BSFL were conducted in Europe (79%), with a few studies originating from Africa and China (Table 2.1). The regulatory landscape, particularly in Europe, restricts food waste as a feed material for BSFL due to concerns about potential contamination [57]. Although agri-food residues are permissible, this limitation may hinder the widespread adoption of BSFL as a feed source. In contrast, regions with less stringent regulations, such as parts of Africa, may offer more favourable conditions for commercializing BSFL-based products. However, the broad adoption of BSFL production has yet to establish a stable market due to factors, such as inconsistent demand, limited infrastructure, and uncertain pricing. In Europe, current regulatory constraints have resulted in a primary focus on BSFL for waste management, as the market for fresh and dried larvae remains underdeveloped. Regarding temporal scope, all the reviewed LCA studies were published within the last decade, reflecting the recent surge in interest in insect-based solutions. Most studies were conducted at the experimental or pilot-scale level, aligning with the developing nature of the insect industry.

**Table 2.1.** LCA studies on Hermetia illucens.

| Index | Author (s) | Year | Country | Type of LCA | Goal of the study |
|---|---|---|---|---|---|
| 1 | Komakech et al. [58] | 2015 | Uganda | Attributional | Comparing the environmental impacts of different biowaste treatment technologies |
| 2 | Smetana et al. [32] | 2016 | Germany | Attributional | Assess the impacts of insect production and processing at the industrial scale |
| 3 | Rustad [59] | 2016 | Norway | Attributional | Compare the environmental impacts of insect meal with fish meal |
| 4 | Salome et al. [22] | 2017 | Italy | Attributional | Assess the impacts of mass-rearing and food waste bioconversion by BSFL in a pilot plant |
| 5 | Mondello et al. [60] | 2017 | Itlay | Attributional | Compare the environmental impacts of different food waste treatment scenarios |
| 6 | Mertenat et al. [24] | 2019 | Switzerland | Attributional | Compare the GWP of BSF biowaste treatment and composting |
| 7 | Smetana et al. [33] | 2019 | Germany | Attributional | Assess the impact of production stages of insect products and compare to benchmarks |
| | | | | Consequential | Identify the environmental consequences of production and consumption choices towards insect-based feed and food |
| 8 | Spinelli et al. [61] | 2019 | Italy | Attributional | Evaluate the impacts of laboratory-scale production of innovative bioplastics made from biopolymers derived from BSF proteins |
| 9 | Bosch et al. [62] | 2019 | Netherlands | Attributional | Assess the environmental burdens of producing fresh BSF larvae reared on different organic biomass resources |
| 10 | Rosa et al. [36] | 2020 | Italy | Attributional | Assess the environmental impacts of BSF protein-derived bioplastic from two protein extraction protocols |
| 11 | Ites et al. [28] | 2020 | Germany | Attributional | Determine the environmental impact of using BSFL to treat food waste in a modular system |
| 12 | Guo et al. [25] | 2021 | China | Attributional | Assess the environmental impact of a BSFL bioconversion plant |
| 13 | Pahmeyer et al. [63] | 2022 | Germany | Attributional | Assess the environmental impact of an automated modular biowaste treatment system for protein production and compare it to conventional sources |
| 14 | Chineme and Assefa [27] | 2023 | Tanzania | Attributional | Determine the environmental impact differences between an open system and closed system BSF waste treatment systems |
| 15 | Beyers et al. [64] | 2023 | Belgium | Consequential | Compare the environmental impacts of different insect diets to produce BSFL<br>Compare the environmental impacts of BSF and soy meal or fishmeal as a feed protein substitute<br>Compare the environmental impacts of BSF composting and conventional waste treatment |

2..3.1.3 System boundaries

BSFL production system can consist of several operational phases due to the many potential products that can be obtained, each of which has its production process. Figure 2.2 shows a general representation of the life cycle stages and system boundaries through which different materials, energy, emissions, and waste associated with BSFL activities and products flow. Generally, the upstream processes include collecting the agri-food residue materials, transporting the substrate to the plant, pre-treating the substrate, and establishing the insect nursery and egg hatchery. The core processes include the bioconversion activities of the larvae, as well as harvesting and processing of the larvae. Minimal processing includes simple operations such as drying and milling the larvae, which can be further processed and refined using various extraction methods to obtain other high-end products. The downstream processes may also include packaging, transportation, use phase, and waste management.

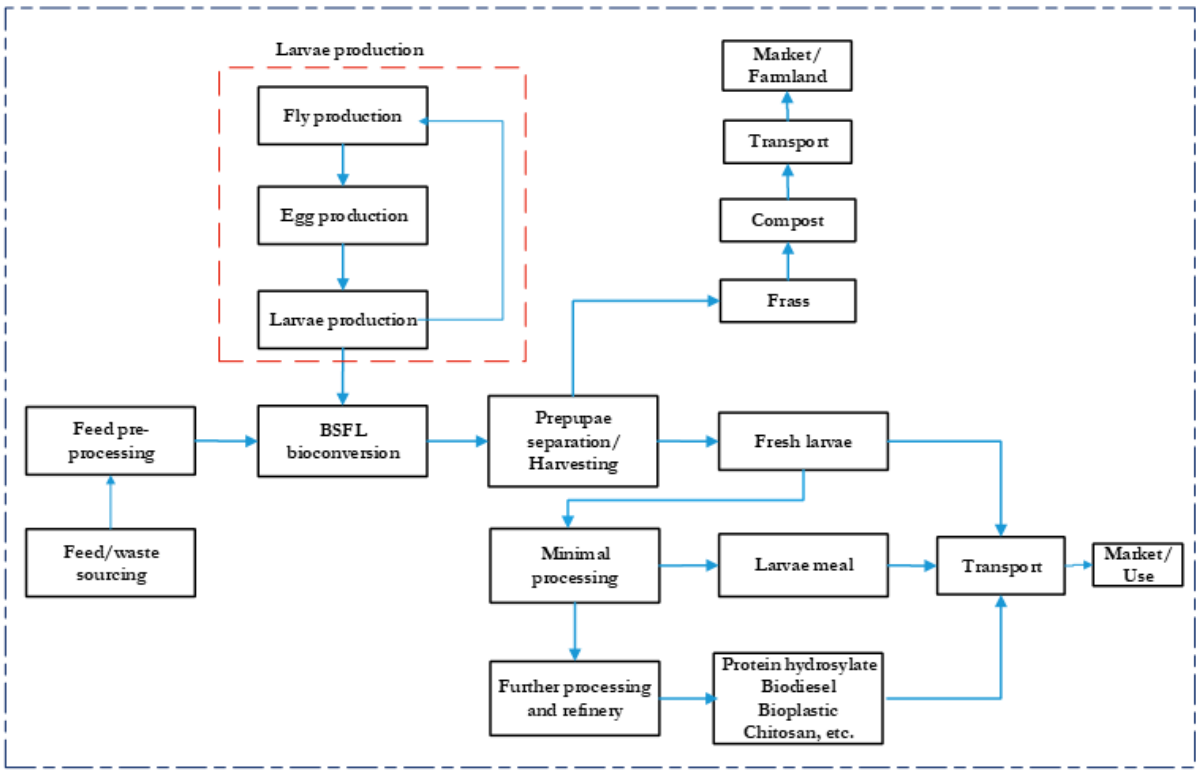



**Figure 2.2.** A generic system boundary including the various life cycle stages of BSFL production and related products. The dotted red lines refer to the young larvae production.

The LCA studies analysed varied in terms of system boundaries, encompassing different phases of the BSFL value chain (Table 2.2). While most studies included upstream and core processes,

such as substrate collection, transportation, bioconversion, and harvesting, there were variations in the consideration of downstream and auxiliary processes. Some studies included the environmental impacts of constructing and maintaining processing facilities and equipment [28, 58, 59]. The inclusion of direct greenhouse gas and ammonia emissions, excluding biogenic $CO_2$ from the bioconversion phase, was observed in certain studies [24, 25, 32, 33, 64]. Additionally, the transportation of materials to the processing facility was commonly considered. Regarding downstream processes, packaging was often excluded, while Komakech et al. [58] considered the distribution phase for insect products. Frass management typically involves direct field application or further processing into a refined organic fertilizer. Wastewater management practices included treatment and, in some cases, reuse [22].

**Table 2.2.** The major phases considered in the LCA studies analysed.

| Index | Fs | Tr | Fp | Lp | Bb | Lsh | Mp | Fpr | Fm | Wm |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | – | – | x | – | x | x | – | – | x | – |
| 2 | x | – | x | x | x | x | x | x | x | x |
| 3 | x | x | – | – | x | x | x | – | x | – |
| 4 | – | x | x | x | x | x | x | – | x | x |
| 5 | x | x | x | – | x | x | x | – | – | – |
| 6 | – | – | x | x | x | x | x | – | x | – |
| 7 | x | x | x | x | x | x | x | x | x | x |
| 8 | x | x | x | x | x | x | x | x | x | x |
| 9 | x | – | x | x | x | x | – | – | – | – |
| 10 | x | x | x | x | x | x | x | x | x | x |
| 11 | x | – | x | x | x | x | x | – | x | x |
| 12 | – | – | – | x | x | x | x | – | x | – |
| 13 | x | x | x | – | x | x | – | – | – | – |
| 14 | x | x | x | x | x | x | – | – | x | – |
| 15 | x | x | x | x | x | x | x | – | x | – |

X: denotes included in the study, – denotes excluded from the study.
Upstream processes: *Fs* feed/waste sourcing, *Tr* transport, *Fp* feed processing, *Lp* larvae production
Core processes: *Bb* BSFL bioconversion, *Lsh* larvae separation/harvesting, *Mp* minimal processing, *Fpr* further processing and refinery
Downstream processes: *Fm* frass management, *Wm* Wastewater management.

#### 2.3.1.4 Functional unit

Regarding BSFL LCA studies, the choice of FU depends on the specific application and the study's objectives. Since BSFL can serve various functions depending on the intended output, multiple FUs were identified in the LCA studies analysed. While some studies focused on the environmental

performance of BSFL production and its by-products, others centred on the waste management function of BSFL. Some studies included more than one FU to facilitate comparisons with conventional products, as shown in Table 2.3. About 60% of the FUs were related to BSFL production and associated products, while the remaining units were linked to biowaste treatment. Selecting an appropriate FU is crucial for accurately comparing environmental impacts. However, the wide range of functional units employed in different studies can make cross-study comparisons challenging.

2.3.1.5 Allocation criteria

As discussed in section 2.4, allocation is a method used in LCA to distribute environmental burdens, material, and energy flows among the various co-products generated by a single technological activity. In BSFL bioconversion systems, multifunctionality arises from the variety of co-products produced simultaneously. Most reviewed LCA studies addressed this by system expansion, substituting commercial products with similar functions. In some studies, however, mass or economic allocation was applied, as shown in Table 2.3. In five of the eight LCA studies involving allocations, the allocation criteria were based on the economic value of co-products, given the significant differences in their economic values. Mass allocation was applied in other studies, including two that specifically investigated bioplastics derived from BSF protein. These studies assessed the quantity of bioplastic produced from extracted BSFL proteins at a laboratory scale, making mass allocation the most suitable approach [65, 66]. For studies primarily centred on BSFL food waste biotreatment, allocation was unnecessary, as the process was considered an end-phase activity [67].

**Table 2.3.** Functional unit, allocation, secondary data sources, LCA method, and LCA software of the analysed BSFL studies.

| Index | Functional unit | Allocation | Data sources | LCIA method | LCA software |
|---|---|---|---|---|---|
| 1 | 1 t of impurity-free biodegradable waste treated | n.r. | literature | CML 2002 | n.s. |
| 2 | 1 kg of dried defatted insect powder produced | mass | Ecoinvent | ReCiPe | SimaPro |
| | 1 kg of ready-for-consumption fresh product | economic | literature | IMPACT 2002+ | |
| 3 | 1 t insect meal produced | economic | Ecoinvent | ReCiPe | Arda Calculator |
| 4 | 1 t of bio digested food waste | economic [a] | Ecoinvent | CML 2 baseline 2000 | SimaPro |
| | 1 kg of proteins produced | | literature | IPCC 2007 | |
| | 1 kg of lipids produced | | | CED and CML 2001 | |
| 5 | 1 t of food waste to be treated | n.r. | Ecoinvent | CML 2 baseline 2000 | SimaPro |
| | | | | CED and CML 2001 | |
| 6 | 1 t of biowaste (ww) treated | n.r. | Ecoinvent | IPCC 2013 | SimaPro |
| 7 | 1 kg of dried and pelletized organic fertilizer produced | economic | Agri-footprint | IMPACT 2002+ | SimaPro |
| | 1 kg of fresh BSF biomass produced | | Ecoinvent | | |
| | 1 kg of protein-concentrated meal produced | | literature | | |
| | 1 kg of BSF fat produced | | | | |
| 8 | 0.403 g of bioplastic produced from BSFL proteins | mass | Ecoinvent | IMPACT 2002+ | SimaPro |
| 9 | 1 kg of fresh larvae produced | economic | Ecoinvent | CML | n.s. |
| | 1 kg of larval protein produced | | literature | | |
| 10 | 0.403g of bioplastic produced from BSFL proteins | mass | Ecoinvent | IMPACT 2002+ | SimaPro |
| 11 | 1 t of treated food waste | n.r. | Eco-invent | IMPACT 2002+ | n.s. |
| 12 | 1 t of treated food waste | n.r. | Ecoinvent | n.s | n.s. |
| 13 | 1 kg of dried larvae and 1 kg of protein produced | n.r | Ecoinvent | IMPACT 2002+ | SimaPro |
| 14 | 1 t of treated food waste | n.r. | Ecoinvent | ReCiPe | OpenLCA |
| 15 | 1 t dry matter BSFL produced | n.r. | Ecoinvent | Environmental Footprint | OpenLCA |
| | 1 t protein contained in livestock feed | | | | |
| | 1 t of bio-digested fresh matter agro-residue | | | | |

*n.r.* not required, *n.s.* not specified, *ww* wet weight, *BSF* black soldier fly, *CED* Cumulative Energy Demand.
* Only for eutrophication.
[a] Does not apply to 1 t of bio-digested food waste.

*2.3.2 Inventory data for BSFL production and biowaste treatment*

In the context of BSFL LCA studies, majority of studies have relied on a combination of primary and secondary data. While primary data from industrial-scale operations would provide the most accurate representation of the system, the limited availability of such data necessitated the use of secondary data and assumptions. This can introduce uncertainties into the analysis, particularly when scaling up from pilot-scale to commercial-scale operations. Approximately 27% of the studies used primary data from industrial sources [24, 32, 33, 59], while others relied on experimental or pilot-scale data with low production volumes. Some studies used no primary data, leading to numerous assumptions and data extrapolations for real-scale production, which may inaccurately reflect environmental performance [28, 63]. It is essential to carefully consider the geographical, temporal, and technological context of the data used in the LCA. Data from different regions or time periods may not be directly applicable to the specific system being studied [68]. Additionally, technological differences can significantly impact the environmental performance of production processes. To improve the accuracy and reliability of future LCA studies on BSFL, it is crucial to prioritize the collection of high-quality primary data. Independent data sources are needed to enhance the credibility and practical relevance of LCA studies on BSFL.

Many LCA studies on BSFL lacked complete inventory data. A significant challenge in conducting accurate LCAs of BSFL production systems is the lack of reliable primary data for key processes, such as larval production, machinery operation, and direct greenhouse gas (GHG) emissions. Direct measurements of GHG emissions from BSFL bioconversion are relatively scarce. Most of the studies did not account for direct GHG emissions in inventory modelling, often due to unreliable background data, while others assumed emissions were negligible [22, 59]. Although some studies have attempted to quantify emissions, variations in experimental conditions and analytical methods have led to inconsistencies in the results [69–76]. Additionally, using secondary data from these sources can lead to inaccurate emission estimates, increasing data uncertainty and reducing result reliability, as the conditions under which the gases were produced may differ significantly. Some studies even relied on direct emissions data for other insect species [22, 32, 33]. However, the use such data may not accurately represent the emissions profile of BSFL systems.

To improve the accuracy and reliability of future LCA studies, it is essential to prioritize primary data collection and develop standardized methods for measuring GHG emissions from BSFL production. By addressing these data gaps and methodological challenges, researchers can provide more robust and informative assessments of the environmental impacts of BSFL-based systems.

### *2.3.3 Life Cycle Impact Assessment*

The impact assessed related to the BSFL production are reported in the section below.

#### 2.3.3.1 Environmental impact coverage

Global Warming Potential (GWP) over a 100-year horizon emerged as the most frequently assessed impact category in the analysed LCA studies, reflecting the global focus on mitigating climate change (Figure 2.3). Over 70% of the studies also evaluated the impacts of energy use, land use, acidification, and eutrophication. Water use was less frequently assessed because insect rearing requires relatively little water. However, it remains an important consideration, particularly in regions with limited water resources. Although focusing on a few key impact categories may be common, a comprehensive LCA should consider a broader range of environmental impacts. Even minor impacts could contribute significantly to the overall environmental impacts, particularly when scaled up. For instance, while often overlooked in BSFL production, water use can become a critical factor in water-scarce regions. By considering multiple impact categories, decision-makers can make informed choices that balance various environmental concerns. However, it is essential to note that a product performing well in one category may underperform in others. Therefore, a holistic assessment of all relevant impact categories is crucial for a comprehensive understanding of environmental performance. Single-score metrics, such as those based on damage-oriented or endpoint impact categories like climate change, ecosystem quality, human health, and resource depletion, can provide a simplified overview [32, 36, 61]. However, they introduce more uncertainties and value judgments, which can affect their accuracy and reliability.

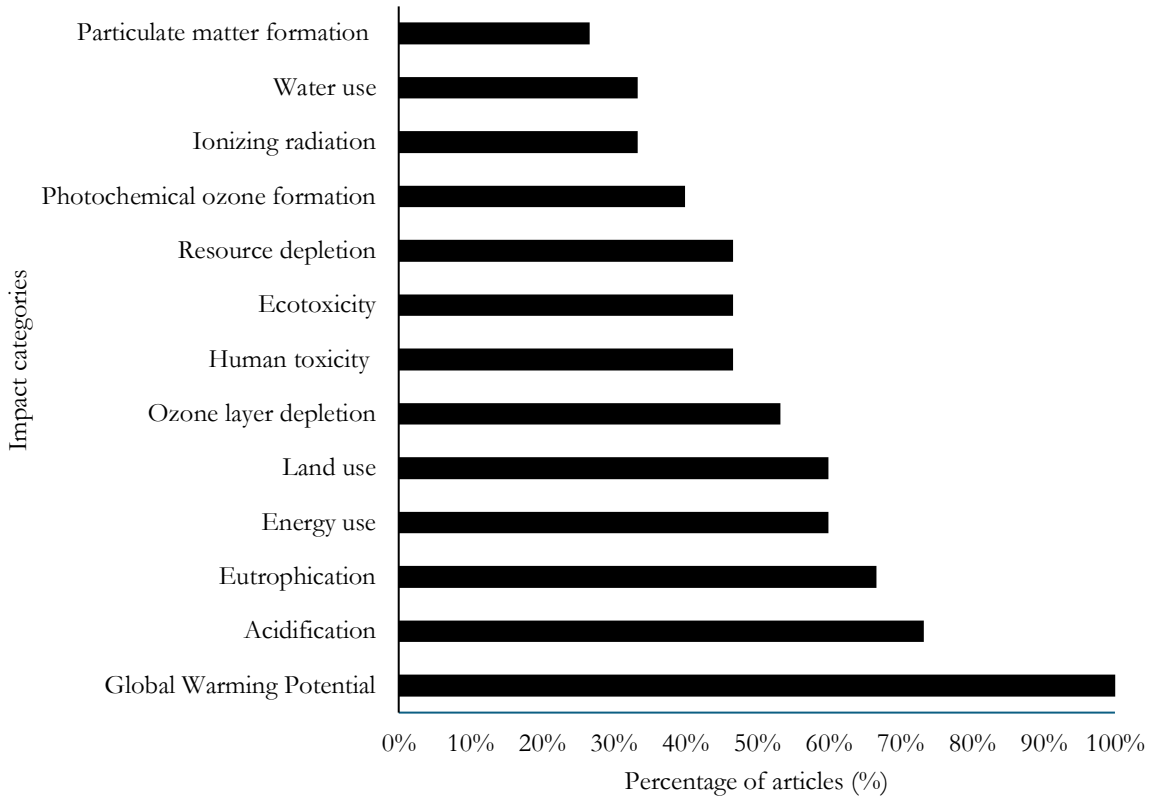



**Figure 2.3.** Frequency of impact categories assessed by the studies as a percentage of the articles that report each impact category.

2.3.3.2 Global warming potential of biowaste treatment by BSFL

Figure 2.4 presents the GWP results from LCA studies focused on BSFL biowaste treatment, expressed per tonne of biowaste treated. The results varied significantly depending on the system boundaries and methodological choices. Studies with narrower system boundaries, excluding processes like substrate collection, transportation, and larvae, reported lower GWP scores [25, 58]. In contrast, studies with broader system boundaries, encompassing additional phases like facility construction and extensive product processing resulted in higher GWP scores [27] as expected.

Although direct GHG emissions from BSFL bioconversion are generally low, indirect emissions, particularly those associated with energy consumption and substrate sourcing, can significantly contribute to the overall environmental impact. Several studies have highlighted the importance of considering both direct and indirect emissions. For instance, Mondello et al. [22] found that electricity consumption and direct emissions from the bioconversion phase accounted for a substantial portion of the total GWP. Similarly, Salamone et al. [22] identified transportation and

processing phases as significant contributors to the environmental impacts. Mertenat et al. [24] attributed 72% of the total GWP to direct emissions from the BSFL treatment facility and 19% to indirect emissions from electricity consumption. Guo et al. [25] also reported direct emissions and electricity consumption as the key hotspots to the total GWP score. At the same time, Chineme and Assefa [27] attributed more than 80% of the impacts to infrastructure.

Including avoided products in LCA studies can significantly influence the overall environmental assessment. By accounting for the environmental benefits associated with substituting conventional products with BSFL-derived products, such as fertilizers and waste treatment, the potential for positive environmental impacts can be highlighted. However, the choice of avoided products and the associated assumptions can significantly impact the results. Moreover, considering the specific context and system boundaries when interpreting LCA results is crucial. Factors such as the type of feedstock, processing methods, and regional differences can influence the relative importance of various impact categories. Additionally, the maturity of the technology and the availability of accurate data can affect the reliability of the assessments.

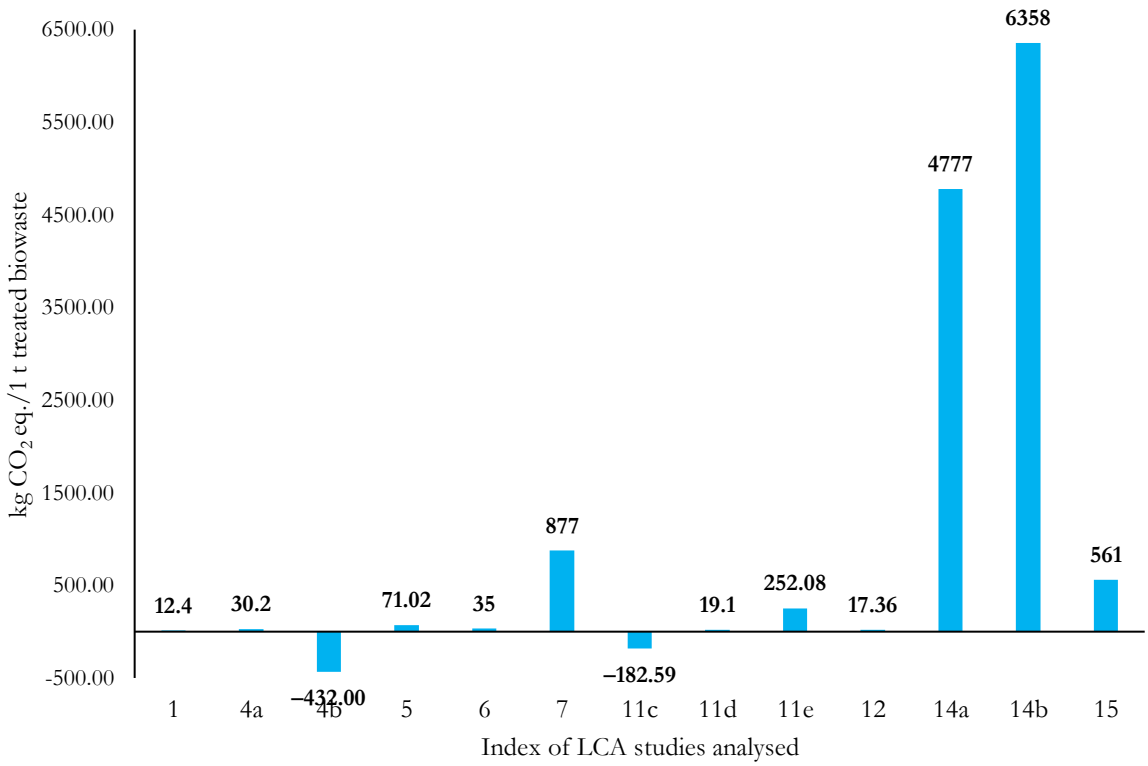



**Figure 2.4.** Global warming potential over a 100-year horizon results reported in LCA studies on biowaste treatment using the BSFL. 4a is without avoided products, 4b is with avoided products, 11c is brewery grains as substrate, 11d is expired food as substrate, 11e is potato peels as substrate, 14a is open system, and 14 b is closed system.

2.3.3.3 GWP for insect products

The LCA studies analysed in this review reported GWP scores for various BSFL products, including fresh larvae, insect protein, and lipids, as shown in Table 2.4. The reported GWP scores varied significantly and were influenced by factors such as the specific production process, diet source, infrastructure, and inclusion of avoided products. Generally, higher GWP scores were associated with refined products of higher economic value, such as protein and lipid extracts [65, 66]. Protein was the predominant insect product of interest in most of the LCA studies analysed due to the promising potential to substitute some conventional protein sources. The primary contributors to GWP included feed production, collection, transportation, feed pre-treatment, biomass processing, and biorefining. While some studies reported lower GWP scores for BSFL protein than conventional sources like egg protein concentrate, microalgae, and whey concentrate [67, 77], others found that BSFL protein had a higher environmental impact than fish meal and soybean [63]. Similarly, Salomone et al. [22] also reported BSFL lipids to have a higher GWP than rapeseed oil.

The choice of feedstock can also influence the environmental performance of BSFL production. Utilizing agro-industrial residues as feed can reduce the environmental impact, but it may also affect the quality and yield of the final product. Bosch et al. [62] reported a lower GWP score for proteins obtained from BSFL reared on residual materials than edible food products for humans. Additionally, the extraction methods used to obtain specific products can significantly impact their environmental impacts. While potentially more efficient, chemical extraction methods may have higher environmental impacts than enzyme-assisted methods [36].

However, it is important to note that many of these studies were conducted at the experimental or pilot-scale level. Therefore, direct comparisons with well-established, commercial-scale production systems may not be entirely accurate. Additionally, the market for BSFL products is still emerging, making it challenging to fully assess the potential environmental and economic benefits of substituting conventional products with BSFL-derived alternatives.

**Table 2.4.** The global warming potential (GWP) scores for different insect products.

| Index | GWP (kg $CO_2$ eq.) | Functional unit |
|---|---|---|
| 3 | 0.17 | 1 kg of insect meal (wet weight) produced |
| 9 | 1 | 1 kg of fresh larvae produced |
| 7 | 1.16 | 1 kg of insect puree (fresh insect production) |
| 13 | 2.77 [b] | 1 kg of dried larvae produced |
| 15 | 8.50 – 20.56 | 1 kg dry matter BSFL produced |
| 4 | 2.1 | 1 kg of insect protein (dried larvae) produced |
| 7 | 5.3 | 1 kg of insect meal (defatted protein concentrate) |
| 13 | 6.96 [b] | 1 kg of protein produced |
| 2 | 1.36 – 15.1 | 1 kg of insect protein meal produced |
| 9 | 19 | 1 kg of protein (food)[a] produced |
| 9 | 3 | 1 kg of protein (feed)[a] produced |
| 9 | 6 | 1 kg of protein (residue)[a] produced |
| 15 | 23.2 - 77 | 1 kg protein contained in livestock feed |
| 10 | 69.8 | 1 kg of extracted protein (chemical extraction) |
| 10 | 188.4 | 1 kg of extracted protein (enzyme assisted extraction) |
| 4 | 2.9 | 1 kg of insect lipid (dried larvae) produced |
| 8 | 1100 | 1 kg of bioplastic obtained from BSFL protein |

[a] drying phase excluded.
[b] No substitution.
*food* – BSFL reared on products humans can consume, *feed* – BSFL reared on livestock feed, *residue* – BSFL reared on organic material not used as food or feed.

2.3.3.4. Energy use

Energy consumption is a significant environmental concern in BSFL bioconversion systems [70,83]. Table 2.5 presents the energy use impacts reported in various BSFL studies. Higher energy demands are often associated with studies that include downstream processes like drying and other biorefinery operations. Maintaining optimal ambient conditions for BSFL production is important

and varies based on the location's climate. For example, studies in warmer regions like Africa did not require climate control, whereas cooler regions like Europe required heating. Additionally, energy for lighting to support adult BSFL mating and energy-intensive processes like biomass drying and milling substantially contribute to overall energy use. The type of substrate can also significantly impact energy consumption due to high energy use during production [28, 33, 62]. Using residual materials instead of food-grade materials for BSFL rearing can reduce energy consumption by up to 85% [62].

In comparative studies, Komakech et al. [58] found that anaerobic digestion and composting required less energy than BSFL-based compost treatment, while vermicomposting required more. Mondello et al. [22] also reported that BSFL biowaste treatment was more energy-efficient than landfilling, incineration, composting, and biogas production. Transitioning to renewable energy sources could help mitigate some of the energy impacts. Salomone et al. [22] conducted a sensitivity analysis on alternative energy sources for the drying process, revealing that photovoltaic and natural gas sources were more efficient than the baseline Italian electricity mix. Similarly, Smetana et al. [33] found that using photovoltaic energy could reduce impacts by 25% in the long term. Thus, adopting renewable energy sources in BSFL facilities is crucial for reducing energy consumption impact.

**Table 2.5.** Impact results for energy and land use reported in the analysed LCA studies.

| Index | MJ | $m^2$ arable | FU |
|---|---|---|---|
| 1 | 15.12 | – | 1 t of food waste treated |
| 4 | 215.30 | 0.67 | 1 t of food waste treated |
| 5 | 772.62 | 0.89 | 1 t of food waste treated |
| 11 | 2846.94* | -1.16 | 1 t of potato peels treated |
| 11 | -23.49* | -0.61 | 1 t of expired food treated |
| 11 | -3067.66* | -477.20 | 1 t of brewery grains treated |
| 14 | 31401 | 1233 | 1 t of food waste treated (open system) |
| 14 | 41491 | 1687 | 1 t of food waste treated (closed system) |
| 15 | -529 – 142 | – | 1 kg of organic waste treated (fresh weight) |
| 7 | 17.90 | 0.48 | 1 kg of puree (fresh insect production) |
| 9 | 8.00 | 2.00 | 1 kg of fresh larvae |
| 13 | 55* [b] | 0.68 [b] | 1 kg of dried larvae |
| 15 | 294.5 – 426.7 | – | 1 kg of dried larvae |
| 2 | 21.20 – 99.60 | 0.03 – 7.03 | 1 kg of insect protein meal |
| 4 | 15.10 | 0.47 | 1 kg of protein (dried larvae) |
| 9 | 174.00* [a] | 67.00 [a] | 1 kg of protein (food) |
| 9 | 84.00* [a] | 3.00 [a] | 1 kg of protein (feed) |
| 9 | 26.00* [a] | – | 1 kg of protein (residue)[a] |
| 13 | 139* [b] | 1.70 [b] | 1 kg of protein |
| 15 | 1120 - 1630 | – | 1 kg protein contained in livestock feed |
| 10 | 15.06 | 0.12 | 1g of extracted protein (chemical extraction) |
| 10 | 40.26 | 0.33 | 1g of extracted protein (enzyme-assisted extraction) |
| 4 | 20.80 | 1.38 | 1 kg of lipid (dried larvae) |
| 7 | 61.29 | 1.38 | 1 kg of lipid |
| 7 | 13.00 | 0.47 | 1 kg of insect fertilizer (dried and pelletized) |

*Results relate to only primary energy.
[a] drying phase excluded.
[b] No substitution.
*food* – BSFL reared on products humans can consume, *feed* – BSFL reared on livestock feed, *residue* – BSFL reared on organic material not used as food or feed.

#### 2.3.3.5 Land use

Land use change and land occupation are two primary aspects of land use that can have significant environmental consequences. Land use change, transforming one land use type into another, can lead to biodiversity loss and habitat destruction. Land occupation, the continuous use of a land area for a specific purpose, can also impact ecosystems and resource availability [78, 79].

In the context of food production, land use is a critical factor. The increasing demand for food has driven the expansion of agricultural land, often at the expense of natural ecosystems. Insect-based protein production, including BSFL, offers a more land-efficient alternative to traditional livestock farming. By utilizing food waste and other organic residues, BSFL can reduce the pressure on agricultural land and minimize deforestation. Table 2.5 summarizes land use impacts from various LCA studies on BSFL; which have shown that BSFL-based systems can have lower land use impacts compared to conventional protein sources, mainly when fed on food waste or other organic residues [67, 80]. Moreover, land use impacts are lowest when BSFL are used for biotreating food waste, while refined BSFL products, such as proteins and bioplastics, have higher land use scores. Some studies reported net positive land use scores when BSFL biotreatment avoided waste treatment needs for organic residue [28, 64]. Furthermore, BSFL biotreatment requires less land than landfills or incineration for organic waste disposal [167]. However, the specific land use impacts can vary depending on factors such as the feeding material used, the scale of production, and the processing methods employed.

### *2.3.4 Sensitivity and uncertainty analyses*

While sensitivity and uncertainty analyses have been conducted in some BSFL LCA studies, their application remains limited. Many studies have relied on simplified assumptions and have not fully explored the range of uncertainties associated with input data and modelling choices. Of the reviewed BSFL studies, 60% included sensitivity analysis, with scenario analysis being the primary method used across all studies. Scenario analysis here refers to assessing the impact of different modelling choices, such as the inclusion of direct and indirect emissions [22], feed inputs [32, 33, 64], substituted products [24], energy consumption [22, 24, 33, 64], and infrastructure and machinery [22, 27]. Additionally, three studies used perturbation analysis, where sensitivity ratios were calculated to identify parameters with significant influence on overall results. All input parameters were varied [24, 25] while others focused on some key parameters such as transport distance and total electricity consumption [22]. In contrast, only one BSFL study included an uncertainty analysis using Monte Carlo simulations to assess parametric data uncertainty [33].

The lack of sensitivity and uncertainty analyses to address data quality limitations, explore relevant scenarios, and account for knowledge gaps in BSFL production and bioconversion activities is a critical gap in LCA studies of BSFL. It is recommended that, at a minimum, sensitivity and scenario analyses be conducted and transparently documented. Where feasible, quantitative uncertainty analyses, e.g., Monte Carlo simulations, should also be included to enhance result reliability. By addressing these limitations and adopting more rigorous approaches, the reliability and robustness of LCA studies of BSFL production systems will be improved.

## 2.4 GREENHOUSE GASES AND AMMONIA ASSOCIATED WITH BSFL PRODUCTION

As mentioned in Section 2.6, BSFL breeding and bioconversion activities emit GHGs and ammonia, which can impact the overall environmental performance of BSFL products. Various factors influence the type and concentration of GHG emissions during BSFL rearing, including substrate quality, experimental design, ambient conditions, sampling procedures, and gas detection and measurement methods, leading to variations between studies (Table 2.6).

Substrate quality is paramount, as it affects the growth rate and biomass quality of BSFL and total GHG emissions. Factors such as dietary composition, volume, moisture content, sensory characteristics, chemical properties, rheology, porosity, and pre-treatment influence substrate digestibility, larval mobility, and the survival and proliferation of microorganisms, all of which can affect GHG emissions. Experimental parameters also play a role; these include the age and density of larvae, feed rate, feeding strategies (e.g., continuous, intermittent, or bulk feeding), and the design of the housing chamber [81]. Additionally, external factors like temperature, humidity, ventilation rate, and light intensity impact both the metabolic activity of BSFL and microbial activity, which can further influence GHG emissions [82, 83].

### *2.4.1 Carbon dioxide*

Carbon dioxide ($CO_2$) is the primary GHG emitted during BSFL rearing. It is primarily produced through the respiration of BSFL and other microorganisms present coupled with the biodegradation of organic matter [69, 71, 72]. While some of this $CO_2$ is derived from biogenic sources, its release can impact the overall carbon balance. A large part of biogenic $CO_2$ emissions comes from the aerobic decomposition of organic matter and anaerobic processes or oxidation of $CH_4$ by aerobic methanotrophic bacteria [84]. While biogenic $CO_2$ is often excluded from LCA assessments, measuring its emissions can provide insights into the carbon cycling and efficiency

of the bioconversion process [71, 85, 86]. By converting organic waste into valuable products, BSFL can mitigate the release of $CO_2$ that would otherwise occur through processes like composting or landfilling. Compared to traditional waste management practices, BSFL-based systems offer a more natural approach to carbon sequestration in the form of insect biomass and reduce the release of GHGs into the atmosphere [85].

Several studies have measured $CO_2$ emissions during BSFL rearing and bioconversion, reporting values ranging from 1 to 4 kg $CO_2$ eq./kg of dry matter BSFL biomass for most substrates (Table 2.6). However, fruit and vegetable waste-based substrates may exhibit higher emission rates. $CO_2$ emissions typically follow a distinct pattern: an initial low phase, a peak during the mid-treatment period, and a gradual decline. This pattern is influenced by substrate quality, larval density, feeding strategies, and ambient conditions. While the impact of these factors on $CO_2$ emissions has been partially explored, further research is needed to understand their influence fully. Non-biogenic $CO_2$ emissions, primarily from energy consumption, also contribute to the overall carbon footprint of BSFL production [67]. The specific emissions depend on factors such as the energy source, efficiency of equipment, and operational practices [76, 84]. To minimize these emissions, optimizing energy use and considering adopting renewable energy sources is essential.

### *2.4.2 Methane*

Methane ($CH_4$), a greenhouse gas, is primarily produced by methanogenic archaea that deoxidize $CO_2/H_2$ and acetic acid under anaerobic conditions [87–89]. While BSFL rearing typically occurs under semi-aerobic conditions, which limits methane production, multiple studies have reported low levels of $CH_4$ emissions (Table 2.6). Although the gut microbiome of the BSFL has not been found to generate $CH_4$ [90], several factors can influence $CH_4$ emissions during BSFL bioconversion, including substrate quality, larval density, and environmental conditions. The early stages of the process, when anaerobic conditions may prevail in some regions of the substrate, can lead to higher $CH_4$ emissions. However, as the BSFL mature and their bioturbation activities improve aeration, $CH_4$ production tends to decrease [24, 91]. Factors influencing the concentration of $CH_4$ emitted during BSFL biotreatment, such as the substrate's nutritional quality, moisture content, and pH, have been summarized in Table 2.7. However, further research is needed to understand the mechanisms driving $CH_4$ production in BSFL systems fully, and to optimize operating conditions to minimize emissions.

**Table 2.6**. A review on quantifying greenhouse emissions per kg of dry matter BSF larvae biomass during rearing and bioconversion.

| Substrate | Larvae age (days) | Larvae density (larvae/cm²) | Feeding rate (mg dm/larva/day) | $CO_2$ (kg/kg dm BSFL) | $CH_4$ (mg/kg dm BSFL) | $N_2O$ (mg/kg dm BSFL) | $NH_3$ (g/kg dm BSFL) | References |
|---|---|---|---|---|---|---|---|---|
| Chicken feed | 7 – 10 | – | 12.61 | 2.2 ± 1.30 | Negligible | – | – | [92] |
| Food waste | 5 | 2 | 30.00 | 1.75 ± 0.17 | 49 ± 29 | 21 ± 13 | – | [73] |
| Kitchen waste | 5 | 4 | 26.00 | – | 5.5 | 118 | – | [75] |
| Pig manure + corn cob | 3 | 0.34 | 38.38 | 2.73 | 455.71 | 1.52 | 8.35 | [72]* |
| Food waste + rice straw | 3 | 0.64 | 23.30 | 1.39 ± 0.34 | 14 ± 6 | 7 ± 1 | – | [71][a*] |
| Pig manure + corn cob | 3 | 1.2 | 15.65 | 1.59 ± 0.13 | 5411.58 ± 3655 | 13.79 ± 3.46 | 7.32 ± 5.02 | [69][b*] |
| Meat and bone meal + rice bran | 3 | 1.13 | 26.67 | 3.33 ± 0.37 | 348.35 ± 390.08 | 6.52 ± 3.85 | 0.11 ± 0.02 | [70]* |
| Yeast concentrates from wheat + starch-rich by-product | 7 | 6.6 | 16.57 | 2.75 ± 0.31 | 19 ± 10[c] | 53 ± 27 | – | [93] |
| Pig manure | 7 | 4.7 ± 0.5 | 22 ± 2 | 344 ± 43 | 10066 ± 2652 | 6 ± 14 | – | [94] |
| Food waste | 4 | 17.95 | 67.85 | 1.17 | 64.31 | 0.33 | 2.90 | [76] |
| Orange peels | – | 6.25 | 7.75 | 5.83 – 34.04 | 22.71 – 208.43 | 3.02 – 24.16 | 0.00 – 18.06 | [74] |
| Broccoli + cauliflower trimmings | – | 6.25 | 3.07 | 3.77 – 22.71 | 561 – 805.86 | 46.35 – 1904 | 4.33 – 125.27 | [74] |
| Food waste | – | 6.25 | 15.00 | 1.36 – 2.70 | 8.27 – 12.77 | 8.27 – 12.77 | 8.27 – 56.56 | [74] |

Calculations and assumptions: For Ermolaev et al. (2019), we used the values of treatment "L."

For Chen et al. (2019), we used 75% moisture content. For Pang et al. (2020[a]), we used the values of treatments pH 5, pH 7, and pH 9.

For Pang et al. (2020[b]), we used the average of all five C/N ratio treatments. For Zhang et al. (2021), we used values for all five treatments.

For Guo et al. (2021), we used the primary results for $CH_4$ and $N_2O$ obtained in the study and not the secondary data in the LCI table

[c] We used mean and standard deviation values without including the outlier reported in their study.

* We assumed the dry matter content of the larvae to be 35%.

**Table 2.7.** The effect of some substrate parameters on the cumulative greenhouse gas emissions and ammonia during BSFL biotreatment.

| Substrate parameter | Carbon dioxide ($CO_2$) | Methane ($CH_4$) | Nitrous oxide ($N_2O$) | Ammonia ($NH_3$) | Reference |
|---|---|---|---|---|---|
| C/N ratio | Generally, substrates with a higher C/N ratio are rapidly degraded, increasing respiration rate and consequent high $CO_2$ loss during larvae growth. Thus, higher cumulative $CO_2$ emissions were observed among treatments with higher initial C/N ratios, with 15 (108 g $CO_2$ eq./kg substrate) being the lowest and 25 (152 g $CO_2$ eq./kg substrate) being the highest. | The C/N ratio inversely correlates with $CH_4$ emissions due to reduced easily degradable C sources. | Generally, very low emissions were detected. However, a low C/N ratio resulted in higher $NH_3$ emissions. BSFL activity could create a conducive aerobic environment that inhibits denitrification and thus limits $N_2O$ production. | A Low C/N ratio resulted in higher $NH_3$ emissions, suggesting that a high C/N ratio of substrate improves microbial assimilation and thus reduces $NH_3$ emissions. On the other hand, a higher proportion of corn cob as part of the BSFL feedstock improves the water-holding capacity, therefore partly reducing $NH_3$ volatilization. | Pang et al. [69] |
| Moisture content | Total emissions correlated positively with increasing substrate moisture content and the overall BSFL biomass yield. | High substrate moisture content positively correlated with higher $CH_4$ emissions due to low oxygen concentration and the formation of anaerobic zones, favoring the growth of methanogenic bacteria | There is no clear relationship between moisture content and $N_2O$ emissions | A negative correlation between moisture content and $NH_3$ emissions was observed, mainly due to the solubility of $NH_3$ in water and the ability of high moisture content substrates to absorb large amounts of $NH_3$ | Chen et al. [72] |
| pH | Emissions were lowest for extreme pH conditions (pH 3.0 and 11.0) and highest at optimum pH conditions of 5.0 and 7.0. | Decreasing the pH of the feeding material also corresponds to a decrease in $CH_4$ production since acidic conditions inhibit methanogen activity | Relatively higher emissions were associated with lower pH treatments, although no statistically significant differences were obtained. | Neutral to high pH (7.0 – 11.0) resulted in higher cumulative $NH_3$ emissions. | Pang et al. [71] |
| Batch feeding time | Increasing the batch feeding times increased $CO_2$ emissions and BSFL biomass. | The periodic addition of substrate led to an anoxic environment under the substrate surface, increasing $CH_4$ emission. | There is no clear relationship between batch feeding times and $N_2O$ emissions, as no statistically significant difference was observed among the various treatments | Increasing batch feeding times increased the cumulative $NH_3$ | Zhang et al. [70] |

*2.4.3 Nitrous oxide*

Nitrous oxide ($N_2O$) is a potent greenhouse gas with a global warming potential 298 times that of $CO_2$ and contributes to ozone layer depletion [95, 96]. $N_2O$ can form through various microbial pathways during substrate biodegradation, particularly as an intermediate or by-product of nitrification and denitrification [97]. In nitrification, microbes convert ammonium to nitrates under aerobic conditions, while in denitrification, $NO_3^-$ is reduced to $N_2$ in anoxic conditions. Denitrification is the primary source of $N_2O$, involving heterotrophic microorganisms that use $NO_3^-$ as an electron acceptor. Factors such as available carbon, oxygen levels, pH, and temperature influence the activity of nitrifiers and denitrifiers in the substrate [98–100].

Although BSFL bioconversion typically occurs under semi-aerobic conditions, which can limit $N_2O$ emissions, it is still essential to consider the potential for $N_2O$ production, particularly during periods of anaerobic activity. BSFL production is associated with very low $N_2O$ emissions (Table 1), often indistinguishable from ambient air levels. Like $CH_4$, $N_2O$ is emitted under anaerobic conditions. However, BSFL feeding and movement aerate the substrate, which inhibits denitrification and reduces $N_2O$ emissions [96, 101–103]. Moreover, the BSFL gut microbiome does not directly produce $N_2O$ [104], and the role of saprophages in $N_2O$ emissions during decomposition remains unclear. Although a symbiotic relationship between gut bacteria and substrate may influence $N_2O$ production, this has yet to be confirmed [87]. Further research is needed to understand the factors that control $N_2O$ production fully and to identify strategies for minimizing emissions.

*2.4.4 Ammonia*

Ammonia ($NH_3$) is not considered a greenhouse gas, but it can be harmful to human health. Additionally, it can harm the environment by contributing to acid rain and acting as a precursor to secondary pollutants like $N_2O$ [98, 105]. Few studies have measured $NH_3$ emissions during BSFL breeding and bioconversion (Table 2.6). During BSFL bioconversion, ammonia emissions are influenced by factors such as feedstock type, environmental conditions, and microbial activity. Typically, $NH_3$ emissions peak during the initial phases of BSFL rearing and decrease quickly after a few days due to rapid organic degradation and the conversion of $NH_4^+$– N to $NH_3$ [87–89, 106, 107]. To reduce $NH_3$ emissions, strategies such as optimizing feeding regimes and controlling environmental conditions can be employed.

### *2.4.5 Other greenhouse gases*

In addition to $CO_2$, other GHGs, such as carbon monoxide (CO) and nitrogen oxides ($NO_X$), can be emitted during the biodegradation of organic matter. While these gases contribute less directly to global warming, they can indirectly affect climate change by increasing $CH_4$ lifetimes and elevating tropospheric $O_3$ concentrations [98]. Although there is limited research on the specific emissions of CO and $NO_X$ from BSFL bioconversion, it is reasonable to assume that they may be produced in small quantities. CO is emitted during the aerobic decomposition of organic material through physical processes and biological activities. $NO_X$ (NO and $NO_2$) is also released as either by-products or intermediates of nitrification and denitrification processes [98]. The formation of these gases is influenced by factors such as the type of feedstock, environmental conditions, and microbial activity. Secondary pollutants, such as peroxyacetyl nitrate, can form through atmospheric reactions involving $NO_X$ emissions [108]. While $NO_X$ emissions from BSFL bioconversion are likely to be minimal, it is vital to consider their potential contribution to air pollution and human health.

## 2.5 DIRECT GREENHOUSE GAS ASSESSMENT METHODS

Reliable measurement techniques are essential to assess and mitigate GHG emissions from the agri-food sector accurately. Various methods have been developed to quantify $CO_2$, $CH_4$, and $N_2O$ fluxes or emissions into the atmosphere to characterize GHG sources and sinks. Common techniques for measuring GHG emissions in field and lab settings include chamber systems, tunnel methods, eddy covariance, relaxed eddy accumulation, container sampling, adsorbent sampling, and integrated horizontal flux methods [109, 110]. Each method has distinct advantages, limitations, and susceptibility to measurement errors.

### *2.5.1 Chamber systems for measuring greenhouse gas emissions*

The chamber system, especially when coupled with container sampling, is the most common method for monitoring GHG emissions during BSFL rearing (Table 2.8). Since BSFL is typically reared in small containers, it can easily be placed in various chamber systems. These chamber systems are generally classified as closed static, closed dynamic, automated, or open chambers.

The closed static chamber is the simplest and most widely used method, where gases accumulate over time in the chamber headspace for subsequent flux rate calculation. The headspace gas can be extracted for analysis, and fans may be added for air circulation [91, 111]. The closed static chamber is simple, cost-effective, and requires manual gas sampling and analysis. They also do not

require sensitive, accurate, or rapid analytical techniques because the extracted gas contains concentrated target compounds [112].

Closed dynamic chambers improve on the static method by continuously pumping accumulated gas to an external analyser, enabling real-time gas measurements and addressing wide concentration variations in the chamber [25]. However, they can be more complex and expensive to set up and maintain. An automated closed chamber offers further advancement, using a pneumatic lid for automated opening and closing, providing continuous, operator-free gas exchange and accurate, high-frequency measurements [113, 114].

Conversely, open chambers maintain a steady air stream through the chamber, with a gas analyser measuring concentration differences between the inlet and outlet. They simulate natural conditions more closely and can be less affected by artificial chamber effects. While easier to use, they may be more susceptible to external factors due to airflow and pressure differences and wind and temperature fluctuations, limiting their application [81, 115].

The choice of chamber system depends on various factors, including the specific research question, the desired level of precision, and the available resources. By carefully designing and implementing chamber systems, researchers can obtain accurate and reliable measurements of GHG emissions from BSFL bioconversion processes.

*2.5.2 Gas sampling techniques*

Gas samples from chamber systems are often collected in specialized containers for laboratory analysis of target GHGs using chromatography. The choice of sampling technique depends on factors such as the target gases, the desired level of precision, and the specific experimental setup. Careful consideration of these factors is essential to ensure accurate and reliable measurements of GHG emissions from BSFL bioconversion systems.

In BSFL rearing studies, common sampling methods involve extracting gas with syringes and transferring it into glass vials, which may be pre-filled with argon, nitrogen-evacuated, or pre-evacuated for sample preservation [24, 87–89, 91, 111, 116]. Alternatively, gas-tight syringes equipped with Luer locks allow direct transport and storage [92]. Syringe sampling is simple and cost-effective. However, sample volume is limited, and there is a potential for gas leakage and contamination. Sampling bags, tubes, and vacuum pumps have been used in larger systems to collect air from closed dynamic chambers. This allows for capturing more representative concentrations over time, especially in open chambers [110]. Sampling bags are easy to use and

cost-effective, though they risk sample loss due to molecular diffusion and film permeation [117]. Thus, it requires careful handling and storage to prevent contamination. Stainless steel canisters are also robust and reusable options for collecting gas samples. Larger sample volumes can be captured by employing sub-atmospheric or pressurized sampling techniques. However, using canisters requires specialized cleaning, preparation, and analysis equipment, which can increase operational costs [117, 118]. Table 2.8 highlights the sampling techniques used in BSFL gas measurement studies.

## 2.6 ANALYTICAL TECHNIQUES FOR GHG MEASUREMENT

Modern analytical techniques for quantifying GHGs rely on diverse physical and chemical principles and are broadly categorized into chromatographic, optical, and electrochemical methods [110, 119]. Some of these methods have been applied to measure GHG emissions from BSFL rearing. Table 2.8 summarizes the various analytical techniques utilized in BSFL studies.

### *2.6.1. Gas chromatography*

Gas chromatography (GC) is a widely used technique for measuring GHGs in chamber systems due to its efficiency in qualitative and quantitative analyses. It offers high sensitivity and accuracy, making it suitable for detecting and quantifying various target gases, including $CO_2$, $CH_4$, and $N_2O$. GC can be coupled with common detectors such as thermal conductivity detector (TCD), flame ionization detector (FID), electron capture detector (ECD), and mass spectrometry (MS) to target specific GHGs based on factors like sensitivity, stability, and response time [110, 120].

The FID is sensitive, especially for $CH_4$, though it can quantify $CO_2$ and CO after conversion to $CH_4$. It works by decomposing samples in a hydrogen flame to release ions and electrons, with electrical conductivity changes measured for quantification. However, FID is destructive, meaning samples cannot be recovered post-analysis [121]. The TCD uses the thermal conductivity difference between carrier gas and sample gas, commonly measuring $CO_2$ emissions in BSFL biotreatment and sometimes coupled with FID for $CH_4$ detection [88]. ECD are highly sensitive to electronegative compounds like $N_2O$ and can detect ppb or ppt levels, ideal for BSFL studies. ECD requires pre-treatment to remove water vapor and $CO_2$ and generally uses argon or nitrogen as carrier gases [119]. MS is a versatile detector capable of identifying and quantifying various compounds, including GHGs. It is highly sensitive and identifies GHGs by measuring the mass-to-charge ratio of the ionized sample fragments. Though accurate, MS is costly and often not ideal for reactive gases [119, 122].

GC gas samples are typically collected using techniques such as syringe sampling, bag sampling, or canister sampling. Proper sample handling and storage are crucial to maintain sample integrity and prevent contamination. Some challenges of GC include the accuracy of the calibration of the GC system, which is essential for reliable quantitative analysis. Additionally, the presence of other compounds in the sample matrix can interfere with the detection of target gases. Data analysis techniques, such as peak integration and calibration curves, are used to determine the concentration of target gases in the samples. By carefully selecting the appropriate GC detector and optimizing the analytical method, accurate and precise measurements of GHG emissions from BSFL bioconversion can be achieved.

### *2.6.2. Optical sensing*

Optical gas sensors offer real-time monitoring of gas concentrations by detecting changes in light absorption or scattering caused by the interaction of gas molecules with specific wavelengths of light [123]. Optical techniques include infrared spectroscopy, laser-based spectroscopy, and photoacoustic spectroscopy.

#### 2.6.2.1. Infrared spectroscopy

Direct gas flow monitoring can be accomplished by utilizing infrared (IR) and near-infrared (NIR) spectroscopy to quantify the specific vibrational transitions of molecular bonds [123]. Non-dispersive infrared spectroscopy (NDIR) and Fourier-transform infrared spectroscopy (FTIR) are the most common IR-based gas analysers, particularly effective in the IR/NIR spectrum. NDIR measures gas concentration by comparing emitted infrared light to the light detected after it passes through gas particles and an optical filter. Known for high accuracy, fast response, and cost-effectiveness, NDIR is especially suitable for $CO_2$ detection with minimal calibration needs. For BSFL biotreatment, NDIR sensors are widely used to monitor GHG emissions, specifically $CO_2$ and $CH_4$ [81, 92, 111]. FTIR also detects molecules based on their unique vibrations, which are linked to functional groups, after infrared exposure. Operating in the IR and NIR spectral regions, FTIR gas analysers can identify multiple components in real-time, with high sensitivity down to ppb, but it is more complex and expensive than NDIR. Portable FTIR spectrometers are now available for continuous field analysis. However, FTIR is limited to measuring gases with nonlinear molecules and can be affected by atmospheric gases like water vapor and ozone. Although FTIR has been applied to measure GHGs such as $CO_2$ and $N_2O$ in agricultural soils and animal production [124], it has yet to be used in BSFL biotreatments.

#### 2.6.2.2. Laser-based spectroscopy

Laser-based spectroscopy involves gas analysers that use a laser to measure gas type and concentration by detecting absorption of specific light wavelengths by target molecules. Operating from visible to mid-infrared ranges, these analysers are well-suited for multi-component and trace gas analysis due to their focused spectral range [123, 125, 126]. Common methods include tunable diode laser absorption spectroscopy (TDLAS), quantum cascade lasers (QCLs), and cavity-ring down spectroscopy (CRDS). TDLAS allows controlled wavelength adjustment to probe specific absorption lines of target gases, enabling high sensitivity and fast response [127, 128]. CRDS measures the light decay time in an optical cavity for precise gas sensing, enhancing sensitivity by increasing the effective path length of light within a high-finesse optical cavity [128, 129]. QCL also offers a wide range of tunable wavelengths, enabling the detection of various gases, including those with complex molecular structures. It uses high-intensity, mid-IR tuning for faster, more accurate trace analysis [123, 130, 131]. However, laser-based techniques can be sensitive to feedback light and temperature and remain untested for BSFL biotreatments [123, 132].

#### 2.6.2.3. Photoacoustic Spectroscopy (PAS)

Photoacoustic spectroscopy (PAS) measures gas concentration by detecting the acoustic signal generated when a gas absorbs light and undergoes thermal expansion. When light is modulated, this expansion creates pressure waves that a microphone detects, making PAS highly effective for measuring gas concentrations [133–135]. However, PAS is extremely sensitive to background noise and interference, which can impact accuracy [110].

Although optical gas sensing techniques have been widely used in various fields, their application in BSFL research is still relatively limited. NDIR sensors have been employed to measure $CO_2$ and $CH_4$ emissions from BSFL bioconversion systems. However, the potential of other techniques, such as FTIR, TDLAS, QCL, and PAS, for more comprehensive and sensitive gas analysis in BSFL research warrants further exploration.

**Table 2.8.** Sampling methods and techniques for measuring greenhouse gas and ammonia emissions during BSFL biotreatment.

| Ref. | Sampling method | Sampling procedure and frequency | Study period | $CO_2$ | $CH_4$ | $N_2O$ | $NH_3$ |
|---|---|---|---|---|---|---|---|
| [85] | Closed static chamber/ Syringe | Three times every morning: one immediately (t=0), at 3 minutes (t=3), and at six minutes (t=6) after sealing the jar. | 7 days | NDIR | GC – FID | – | – |
| [73] | Closed static chamber + fan/Syringe | Three days 2, 7, and 11 (5-day intervals). Samples were taken at 0, 15, 30, and 60 min intervals. | 14 days | GC – TCD | GC – FID | GC – ECD | EC gas detector tube system |
| [75] | Closed static chamber + fan/Syringe | Daily in triplicates | 13 days | – | GC – FID | GC – ECD | – |
| [72] | Closed static chamber/ Syringe | Every morning | 8 days | GC – FID | GC – FID | GC – ECD | Trapped in an $H_2SO_4$ solution and then measured by titration with a NaOH solution |
| [71] | Closed static chamber/ Syringe | Daily at 0 and 20 min after the container closure | 10 days | GC – FID | GC – FID | GC – ECD | Tapped in an $H_2SO_4$ solution and then measured by titration with a NaOH solution |
| [69] | Closed static chamber/ Syringe | Daily at 0 and 300 s | 12 days | GC – FID | GC – FID | GC – ECD | Trapped in an $H_2SO_4$ solution and then measured by titration with a NaOH solution |
| [70] | Closed static chamber/ Syringe | Every morning | 10 days | GC – FID | GC – FID | GC – ECD | Trapped in an $H_2SO_4$ solution and then measured by titration with a NaOH solution |
| [93, 94] | Open chamber + fan/ Syringe ($N_2O$ only) | Daily. Every 9 min from the ingoing ($CO_2$ and $CH_4$) and outgoing airstream ($CO_2$, $CH_4$, and $NH_3$). $N_2O$ samples were taken every 24 h (at noon). | 8 days | NDIR | NDIR | GC – ECD | EC gas analyzer |
| [136] | Closed static chamber | Daily at 11:00 h at intervals of 0.10.20,40, and 60 min. | 6 days | – | GC – FID | – | – |
| [76] | Closed dynamic chamber + fan/ Sampling bag and tube | Sampling was done over a three-day period with 4 h intervals (4:00, 8:00, 12:00, 16:00, 20:00, 24:00). | 10 days | – | GC– ns | GC – ns | EC gas detector |
| [74] | Closed static chamber + fan/ Syringe | Samples were taken four times, three between 1 to 7 d and the last between 7 – 25 d depending on the BSFL composting duration (directly upon sealing the box, after 20 min, and after 1 h). | 17 – 35 days | EC gas detector tube system | GC – FID | GC – ECD | EC gas detector tube system |

NDIR – non-dispersive infrared analyzer, GC – Gas chromatography, FID – Flame ionization detector, ECD – Electron capture detector, TCD – Thermal conductivity detector, EC – Electrochemical, Ns – detector not specified.

*2.6.3. Electrochemical sensing*

Electrochemical (EC) sensors offer a promising approach to detecting, quantifying, and monitoring gases, especially in environmental air monitoring. These sensors convert a chemical reaction on an electrode into an electrical signal, often using oxidation-reduction reactions with the target analyte [137]. Key electrochemical methods include potentiometry, conductometry, amperometry, coulometry, and capacitance. Potential differences are measured based on Nernst's equation in potentiometric sensing, while conductometric sensors detect resistance changes due to target gases [137–139]. Amperometry and coulometry measure current as a function of applied potential or time, providing real-time detection [138].

Electrochemical gas sensors usually confer portability, ease of use, low noise, low power consumption, simplicity, high sensitivity, high selectivity, affordability, real-time studies, in-situ, and rapid analyses of target species [137–139]. However, they work best for low gas concentrations and electrochemically active gases. For non-reactive gases like $CO_2$, indirect measurement requires generating a detectable response via a secondary reaction. Other limitations include stability issues, lifespan, and temperature sensitivity for some electrolytes [110, 125]. Commercial sensors now detect various gases at ppm and ppb levels, using materials like polymers, metals, and metal oxides [110, 140]. EC sensors have been widely used to measure $NH_3$ emissions in BSFL production, with systems like the Kitagawa gas detector also applied for $CO_2$ detection. However, using EC sensors to detect other GHGs, such as $CO_2$ and $CH_4$, is less common [25, 81, 91, 111]. Further advancements are needed to develop and optimize EC sensors for the specific requirements of BSFL research, including improved sensitivity, selectivity, and long-term stability.

*2.6.4. Remarks*

Despite recent gas detection and quantification advancements, accurately measuring low-level emissions from BSFL bioconversion remains a challenge. The choice of analytical method, instrument calibration, and the availability of reliable reference standards are critical factors influencing the accuracy of results. Additionally, the complex interplay between various factors, such as feed quality, experimental conditions, and microbial activity, can impact the emission profiles of different GHGs. While some studies have investigated these factors, a more comprehensive understanding is needed to optimize BSFL production systems for minimal environmental impact. In many studies, the lack of real-time monitoring limits our ability to fully characterize emission patterns and identify critical periods of high emission intensity. The high cost of gas sensors and analytical equipment is a significant barrier to real-time monitoring of

GHG emissions, particularly in small-scale laboratory and pilot studies. Implementing continuous monitoring systems would provide valuable insights into the dynamics of gas emissions and enable the development of strategies to mitigate them.

## 2.7. CONCLUSIONS

This literature review highlighted current trends in applying LCA to BSFL production and bioconversion, including the strengths and limitations of methods used to detect and quantify GHG emissions in these systems. There is currently no standardized approach to performing LCAs for BSFL production and related products. Differences in functional units, system boundaries, allocation criteria, and impact assessment methods vary widely. Without a Product Category Rules (PCR) specific to insect products, results across studies lack comparability, as PCRs provide essential guidelines for LCA consistency within similar product categories. Due to the multifunctional nature of the BSFL value chain, establishing a methodological consensus is crucial to ensuring unbiased environmental assessments.

Most studies employed attributional LCA due to its suitability for assessing environmental impacts. However, using consequential LCA approaches could enhance insights into future strategies as both approaches address different relevant questions. A major challenge to LCA transparency and consistency in BSFL studies is data quality within the inventory phase. Significant uncertainties arise when secondary data replaces primary data, limiting accuracy. Many studies, particularly at experimental or pilot scales, rely on secondary data for key inventories, resulting in more generalized and less realistic outcomes. Primary data on energy use, infrastructure, and emissions, particularly at the commercial scale, are essential for accurate impact assessments.

Regarding impacts assessed, the reviewed studies were mostly focused on climate change, often overlooking other impact categories crucial for a holistic environmental evaluation. Including a broader range of impact categories would enable a more balanced comparison between BSFL and conventional products. While bioconversion and processing were generally the most impactful phases, variations in GWP, energy, and land use were largely due to differences in data quality, system boundaries, and assumptions across studies.

Further LCA research on various BSFL products, including biodiesel and edible films, is needed to evaluate the BSFL value chain comprehensively against traditional alternatives. Additionally, no nutritional LCA studies exist for BSFL-based feed for livestock and pets. Given that the primary function of BSFL feed is nutritional, an LCA that links nutritional value with environmental impact

would be valuable. Moreover, there is a need to consider the exploration of social and ethical implications of BSFL production, including issues related to animal welfare, food safety, and consumer acceptance. Although scenario analysis is common, sensitivity and uncertainty analyses (e.g., perturbation and quantitative uncertainty analysis) remain rare. Advancing LCA methodologies in BSFL production and bioconversion is essential to support informed, sustainable feed and biowaste management decisions.

Regarding gas emissions associated with BSFL activities, traditional chromatographic methods have been widely applied to measure gas emissions from BSFL biotreatment, offering accuracy but at a high cost, with labour-intensive processes and no real-time monitoring. This highlights the need for EC sensors, which are affordable, sensitive, easy to install, and capable of real-time monitoring, particularly through wireless data transmission. EC could be effectively deployed to monitor air quality in commercial BSFL facilities. Factors such as electrode type, detection limits, accuracy, and sensor durability should be carefully considered when selecting sensors.

More comprehensive and continuous monitoring studies are needed to gain a deeper understanding of the factors influencing GHG emissions from BSFL bioconversion. Although batch sampling provides valuable information, real-time monitoring can reveal dynamic changes in emission patterns and identify critical periods of high emission intensity. By investigating the impact of factors such as substrate quality, environmental conditions, and operational parameters, researchers can develop strategies to optimize BSFL production systems and minimize environmental impacts. Additionally, developing advanced modelling techniques can help predict GHG emissions based on specific input parameters, enabling more accurate assessments of the environmental impact of BSFL production. Understanding how various parameters impact gas emissions is essential for predicting emission types and volumes, as well as for optimizing BSFL biomass quality and yield. By combining experimental data with modelling approaches, it is possible to identify opportunities for reducing emissions and improving the overall sustainability of BSFL-based systems.

# 3 CASE STUDY (LCA OF FROZEN VEGETABLES)

Part of the contents of this Chapter were published in:

- Sustainability

  *Boakye-Yiadom, K.A., Ilari, A., Bisinella, V. and Duca, D., 2023. Environmental Impact Assessment of Frozen Peas Production from Conventional and Organic Farming in Italy. Sustainability 2023, 15, p. 13373.* ***https://doi.org/10.3390/su151813373***

- Resources, Environment and Sustainability

  *Pedretti, E.F.; Duca, D.; Ballarini, M.; Boakye-Yiadom, K.A. and Ilari, A. Boakye-Yiadom, K.A., Ilari, A., 2023. Environmental Impact Assessment of Producing Frozen Spinach in Central Italy. Resources, Environment and Sustainability* ***2023****, 12, p. 100110.*

  ***https://doi.org/10.1016/j.resenv.2023.100110***

- Sustainability

  *Foppa Pedretti, E.; Boakye-Yiadom, K.A.; Valentini, E.; Ilari, A. and Duca, D., 2021. Life Cycle Assessment of Spinach Produced in Central and Southern Italy. Sustainability* ***2021****, 13, p. 1000*

  ***https://doi.org/10.3390/su131810001***

- AIIA 2022: Biosystems Engineering Towards the Green Deal (Conference Proceedings)

  *Ilari, Alessio., Boakye-Yiadom, K.A, Duca, D. and Foppa Pedretti, E., 2023. Valorizing Agro-Industry Residues to Improve the Environmental Sustainability of Frozen Products 2023. In: Ferro, V., Giordano, G., Orlando, S., Vallone, M., Cascone, G., Porto, S.M.C. (eds) AIIA 2022: Biosystems Engineering Towards the Green Deal. AIIA 2022. Lecture Notes in Civil Engineering, vol 337. Springer, Cham.* ***https://doi.org/10.1007/978-3-031-30329-6_66***

## 3.1. CHAPTER SUMMARY

In the preceding Chapter, the LCA methodology was reviewed and its application to BSFL production. Key methodological challenges and opportunities for improvement in BSFL LCA studies were identified. Building upon these insights, this Chapter presents an overview of the case study to evaluate the environmental performance of different frozen horticultural products, incorporating BSFL biotreatment of associated residues. This Chapter introduces the general LCA framework applied for all the case studies, but the detailed inventory and model specifications are reported in Chapter 4 to 7.

The primary objectives of this case study are to:

i. Assess the environmental impacts of producing frozen horticultural products.
ii. Evaluate the environmental benefits of producing BSFL through the valorization of agro-industrial residues.
iii. Identify key hotspots within the production chain for potential improvements.

To achieve these objectives, the production chain of frozen horticultural products is divided into four chapters:

- Chapter 4 (Cultivation Phase): This part covers crop cultivation, including land use, water consumption, and agrochemicals application.
- Chapter 5 (Processing Phase): This stage encompasses transportation and processing operations, such as washing, cutting, blanching, and freezing.
- Chapter 6 (Residue Valorisation with BSFL): This section focuses on the environmental impacts of producing BSFL as a means to valorise the residue from the processing phase.
- Chapter 7: This section summarizes the overall environmental impacts of the frozen horticultural product system, highlighting the contribution of each stage and the potential benefits of incorporating BSFL biotreatment. Additionally, this section explores the environmental performance of BSFL biotreatment as an alternative waste management to anaerobic digestion.

## 3.2 INTRODUCTION

### *3.2.1 Goal and scope definition of the case study*

This case study examined the environmental impacts of using BSFL biotreatment for agri-food industrial residues as a strategy to enhance the sustainability of frozen horticultural products. The study focused on the activities of an Italian agricultural consortium company, C.O.VAL.M. – O.R.T.O. Verde S.c.a.p.a, which specializes in the production and sale of minimally processed frozen vegetables. The company was founded to create a single, organised supply line that could be directly managed by local growers in order to process vegetables locally. Operating across domestic and export markets, the company engages in both business-to-business (B2B) and business-to-consumer (B2C) transactions, serving diverse market segments. Basil, garlic, chard, broccoli, carrots, cauliflower, chicory, turnip, onion, borlotti beans, green beans, green peas, spinach, tomato, leek, celery, and shallot are some of the produces cultivated mainly under integrated open field system. The company produces about 50,000 t of frozen vegetables annually

from over 6,000 ha of cultivated land, primarily located in central and southern Italy. Its products are supplied to retailers, the food industry, and major frozen vegetable producers across Italy.

However, the production process generates significant waste, up to 30% of the input, consisting of unwanted parts and defective products. Currently, this waste is sent to an agricultural biogas plant for biogas and digestate production. To further valorize the residue and improve supply chain sustainability, a BSFL biotreatment process was proposed. To this end, a pilot plant with a capacity to treat 1.5 t of biowaste per cycle was constructed to manage the residues. Therefore, the primary aim of the study was to compare the environmental performance of BSFL biotreatment for various agri-food residues against the conventional anaerobic digestion (AD) method. Given that the BSFL plant is still in its early stages of testing, the study aimed to optimize the biotreatment process by identifying key parameters and environmental hotspots. A critical objective was to assess potential trade-offs, particularly where environmental benefits from BSFL products might arise through the substitution of conventional products.

This LCA was designed to provide insights for improving the environmental sustainability of frozen horticulture value chains. The findings are intended to benefit C.O.VAL.M. - O.R.T.O. Verde S.c.a.p.a and stakeholders across the horticultural sector, including farmers, processors, waste management companies, environmental authorities, policymakers, researchers, NGOs, and insect enthusiasts. By providing insights into the environmental benefits of BSFL biotreatment, this study also aims to contribute to developing more sustainable food production systems.

#### 3.2.1.1 Function and Functional Unit

The primary function of the system is to produce frozen horticultural products. However, agro-industrial residue management can generate valuable secondary outputs, including BSFL and frass from BSFL biotreatment, as well as electricity, heat, and digestate from anaerobic digestion. To comprehensively assess the environmental implications of the horticultural value chain, including those related to BSFL production, the LCA employed a multi-dimensional approach with five assessments, each defined by a specific functional unit:

- Assessment 1: The environmental impacts of cultivating various horticultural products under integrated and organic production systems.

  - Functional unit: The production of 1 t of freshly harvested matured horticultural product cultivated in Italy between 2019 to 2022.
- Assessment 2: The environmental impacts of producing different horticultural products from organic and integrated farming systems.
  - Functional unit: The production of 1 t of packaged frozen vegetables from the processing of fresh horticultural products between 2019 to 2022.
- Assessment 3: Comparison of the environmental impacts of different agro-industrial residue diets used to produce BSFL.
  - Functional unit: The production of 1 t of fresh BSFL biomass from a semi-industrial scale prototype plant using agro-industrial residues in 2024.
- Assessment 4: Comparison of the environmental impacts of BSFL-derived protein and lipids with conventional products.

  Functional unit:
  - The provision of 1 kg of protein in feed.
  - The provision of 1 MJ of energy from lipids.
- Assessment 5: Comparison of the environmental impacts of BSFL biotreatment versus conventional agro-industrial residue treatment for the same residues.
  - Functional unit: The treatment of 1 t of fresh matter agro-industrial residue in 2024.

The study follows the LCA methodology in accordance with ISO standards. Given the focus on evaluating the environmental performance of an existing system, an attributional LCA approach was adopted. This approach, based on a micro-level perspective, involves modeling the system's current state using average data for processes and inputs. To account for the co-products generated from the agro-industrial residue management, system expansion was employed. This involved crediting the system for the environmental benefits associated with the production of these co-products, such as BSFL biomass and biogas. Average Italian data were used for crediting to ensure consistency and relevance to the study's geographic scope.

The system boundary of the attributional LCA study includes the crop cultivation phase, industrial processing of horticultural products, and BSFL biotreatment of residues (Figure 3.1). Considering that the main products obtained from the BSFL biotreatment were the larvae and frass, which can serve as sources of protein, lipid, and fertilizer, respectively, the system was expanded to allow for substitution with commercial products that perform similar functions.

Therefore the environmental load avoided in the production of these products were considered as environmental credits in the production of 1 t of frozen vegetable or 1 t of BSFL biowaste treatment. Additionally, the avoided products from the AD treatment of the residue was also included, mainly electricity, heat, and mineral fertilizer.

However, some key stages with the horticultural value chain such as transportation to retail, retail, product storage and use by consumer, and packaging end-of-life were excluded due the unavailability of reliable data.

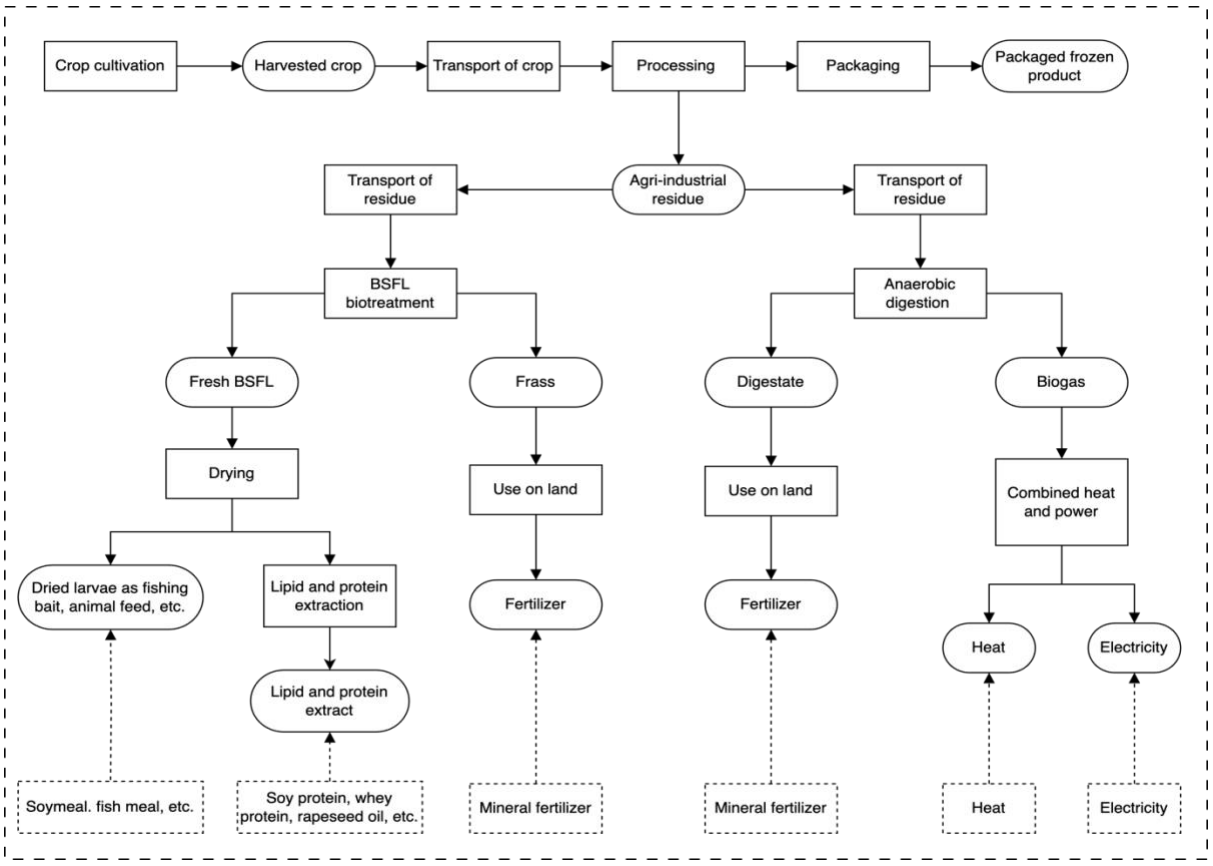



**Figure 3.1**. Process flow diagram of the LCA scenario. Circles are for output products. Squares are for processes. Dashed squares are for energy and material substitution (avoided production).

### *3.2.2 Life Cycle Impact Assessment*

The impacts, per the selected functional units, were evaluated using the Environmental Footprint (EF) 3.1 midpoint life cycle impact assessment (LCIA) method [141] due to its comprehensiveness in terms of environmental issues covered and its scientific soundness. Details of the impact categories, abbreviations, reference units, and description used are reported in Table 3.1.

**Table 3.1**. List of midpoint impact categories with their units and descriptions for the EF 3.1 Method used in the LCA study.

| Impact category | Unit | Description |
|---|---|---|
| Climate change (CC) | kg $CO_2$ eq. | Refers to the increase in the average global temperatures due to GHG emissions. CC is a method for comparing the climate effects of emissions of different GHGs like $CO_2$, $CH_4$, and $N_2O$. |
| Ozone depletion (ODP) | kg CFC-11 eq. | The stratospheric ozone ($O_3$) layer protects us from hazardous ultraviolet radiation (UV-B). Its depletion increases skin cancer cases in humans and damage to plants. |
| Human toxicity, cancer effects (HT_c) | CTUh | Refers to potential impacts, via the environment, on human health caused by absorbing substances from the air, water and soil. Direct effects of products on human health are currently not measured. |
| Human toxicity, non-cancer effects (HT_nc) | CTUh | Refers to potential impacts, via the environment, on human health caused by absorbing substances from the air, water, and soil. Direct effects of products on human health are currently not measured. |
| Particulate matter (PM) | disease incidence per kg $PM_{25}$ emitted | Measures the adverse impacts on human health with exposure to particulate matter (PM) and its precursor (e.g. NOx, $SO_2$). Smaller PM particles, which can penetrate deep into the lungs, pose a greater health threat. |
| Ionising radiation (IR) | kg $U^{235}$ eq. | The exposure to ionising radiation (radioactivity) can have impacts on human health. Only emissions under normal operating conditions are considered. |
| Photochemical ozone formation (POF) | kg NMVOC eq. | Ozone ($O_3$) on the ground (in the troposphere) is harmful: it attacks organic compounds in animals and plants, it increases the frequency of respiratory problems when photochemical smog ("summer smog") is present in cities. |
| Acidification (AP) | mol $H^+$ eq. | Some emissions from anthropogenic activities can reduce pH due to the acidifying effects in water and soil systems. Acidifying substances cause a wide range of impacts on soil, groundwater, surface water, organisms, ecosystems, and materials consequently, decreasing biodiversity and damaging the quality of ecosystems. Gases with high acidifying potential include $NH_3$, (NOx), and (SOx) |
| Eutrophication, terrestrial (TEU) | mol N eq. | Arises when substances containing nitrogen (N) or phosphorus (P) are released to ecosystems. These nutrients cause a growth of algae or specific plants and thus limit growth in the original ecosystem. |

| | | |
|---|---|---|
| Eutrophication, freshwater (FEU) | kg P eq. | Happens when substances containing nitrogen (N) or phosphorus (P) are released to the ecosystem. As a rule, the availability of one of these nutrients will be a limiting factor for growth in the ecosystem, and if this nutrient is added, the growth of algae or specific plants will increase. If algae grow too rapidly, it can leave water without enough oxygen for fish to survive. |
| Eutrophication, marine (MEU) | kg N eq. | Happens when substances containing nitrogen (N) or phosphorus (P) are released to the ecosystem. As a rule, the availability of one of these nutrients will be a limiting factor for growth in the ecosystem, and if this nutrient is added, the growth of algae or specific plants will increase. If algae grow too rapidly, it can leave water without enough oxygen for fish to survive. For the marine environment this will be mainly due to an increase of nitrogen (N). |
| Ecotoxicity, freshwater (ETF) | CTUe | This indicator refers to potential toxic impacts on an ecosystem, which may damage individual species as well as the functioning of the ecosystem. Some substances tend to accumulate in living organisms. |
| Land use (LU) | Pt | Use and transformation of land for agriculture, roads, housing, mining or other purposes. The impacts can vary and include loss of species, of the organic matter content of soil, or loss of the soil itself (erosion). This is a composite indicator measuring impacts on four soil properties (biotic production, erosion resistance, groundwater regeneration and mechanical filtration). |
| Water use (WU) | $m^3$ of deprived water | The abstraction of water from lakes, rivers or groundwater can contribute to the 'depletion' of available water. The impact category considers the availability or scarcity of water in the regions where the activity takes place, if this information is known. |
| Resource use, fossils (FRD) | MJ | The earth contains a finite amount of non-renewable resources, such as fossil fuels. The basic idea behind this impact category is that extracting resources today will force future generations to extract less or different resources. |
| Resource use, minerals and metals (MRD) | kg Sb eq. | This indicator is related to the consumption of non-biological resources such as minerals on a global scale as it measures the scarcity of a substance over time. |

Adapted from Mengual et al. [142] .

# 4 CULTIVATIONS OF HORTICULTURAL PRODUCE

## 4.1 CHAPTER SUMMARY

This part of the study applies LCA to evaluate the environmental performance of pea, spinach, tomato, and onion cultivation by C.O.VAL. M. – O.R.T.O. Verde S.c.a.p.a. To this end, an attributional LCA was conducted using primary data collected between 2019 and 2022 from the company. The Environmental Footprint Method 3.1 was employed to assess midpoint impact categories for integrated and organic cultivation. The average results indicate that integrated production generally had a relatively higher environmental impact than organic production due to the use of synthetic agrochemicals and their associated emissions. However, it is important to note that some fields in organic production had zero yield, which can skew the results. Additionally, due to the larger sample size of integrated farms compared to organic farms, a greater degree of variability in yield was observed within the integrated farming group. Based on all these limitations, it was inconclusive to ascertain which farming system was better. Therefore, selecting the most environmentally sustainable farming system should consider local factors like cultivar, soil, and climatic conditions that influence productivity. Strategies to reduce environmental impacts for the integrated farming could include, reducing fertilizer use improving nitrogen use efficiency through nitrification inhibitors and harnessing the nitrogen-fixing capabilities of crops like peas. By implementing these strategies, the horticultural industry can improve the environmental sustainability of frozen vegetable production and contribute to more sustainable diets.

## 4.2 INTRODUCTION

The agricultural sector significantly contributes to environmental issues such as climate change, land degradation, water pollution, and biodiversity loss [143–145]. A substantial portion of these impacts is linked to food production, which accounts for a considerable share of global GHG emissions and resource consumption [146]. As one of the primary contributors to global environmental issues, the agri-food sector accounts for approximately a third of the total anthropogenic GHG emitted from agriculture, forestry, and other land uses [147]. Climate change poses a significant threat to agriculture, as many crops are susceptible to changes in temperature, precipitation, and extreme weather events [148–150]. Consequently, environmental sustainability within the agri-food sector has gained significant attention due to its critical role in ensuring resilient and sustainable food production systems.

To address these challenges, adopting sustainable agricultural and food consumption practices is crucial. The 2030 Agenda for Sustainable Development recognizes the importance of sustainable food systems and highlights the need to balance food security, nutrition, and environmental protection [151, 152]. Shifting to plant-based diets can reduce the environmental impact of food production and promote health [153, 154]. However, crop production systems can also have adverse environmental impacts [155, 156]. Therefore, gaining a comprehensive understanding of the environmental implications of various crops is essential. This will enable more informed decisions about integrating diverse crops into our diets, thereby improving our food systems' sustainability and nutritional diversity.

Environmental sustainability has become a critical focus in the food sector, driven by growing consumer demand for environmentally responsible practices and regulatory pressures. In response, many actors along the food value chain have initiated efforts to reduce their environmental impacts [157–159]. Accurate and thorough environmental impact assessments are essential to support these efforts, given the complexity of processes and materials involved. The LCA methodology is one of the prominent methods developed to quantify these impacts. LCA is a standardized and comprehensive decision-making tool that evaluates the environmental impacts of a product throughout its entire production system [38, 39]. This analysis helps identify areas for improvement and inform decision-making. LCA is widely applied across the agri-food sector, enabling the assessment of agricultural products cultivated in various production systems and supporting informed decision-making at multiple levels [160, 161].

LCA studies can provide valuable insights into the environmental performance of different food products, enabling comparisons between conventional and innovative production systems. Additionally, LCA can support the development of eco-labeling schemes and sustainable procurement practices [160, 161]. As companies increasingly prioritize green procurement, many are incorporating LCA into their product evaluations and requiring LCA data from suppliers [162]. This growing demand for transparency in environmental information is driving food ingredient suppliers to assess their products, improve efficiency, and minimize environmental impacts, aligning with the sector's sustainability goals [163, 164]. By understanding the environmental implications of food choices, consumers and businesses can make informed decisions that contribute to a more sustainable future.

Given the increasing global demand for vegetables driven by health-conscious dietary trends, cultivating horticultural crops has become a significant agricultural activity [165]. The vegetable sector plays a crucial role in both domestic and international markets within Europe. In 2022, the

value of fresh vegetable production reached EUR 40.7 billion, with Italy contributing 21.3% as the second-largest producer [166]. However, vegetable production can lead to environmental challenges, including land use, water consumption, and GHG emissions [167]. Therefore, this study seeks to quantify the environmental burdens associated with horticultural production by a central Italian supplier, with the goal of identifying environmental hotspots and opportunities for sustainability improvements in the horticulture supply chain.

## 4.3 MATERIALS AND METHODS

To evaluate the potential environmental impacts of the cultivation phase, the ISO 14040/14044 standards [38, 39] and the International Reference Life Cycle Data System (ILCD) Handbook [40] were relied upon to conduct the LCA study. The LCA software SimaPro version 9.5. was used. Background data for ancillary materials and energy were obtained from the Ecoinvent database version 3.9 (allocation, cut-off by classification processes) [168].

### *4.3.1 Goal and scope definition*

The primary objective of this LCA study is to evaluate the environmental impacts associated with cultivating horticultural products (spinach, peas, tomato, and onion) for minimal processing into frozen whole or cut products by C.O.VAL. M. – O.R.T.O. Verde S.c.a.p.a. The study aims to assess the environmental performance of integrated and organic production systems and identify key environmental hotspots within the horticultural production process. However, for tomato cultivation, only the integrated production system was analyzed, as data for organic fields were not available. The specific objectives of this analysis are to:

- establish a baseline environmental performance assessment for the cultivated crops under both integrated and organic production systems.
- assess the environmental impacts of different horticultural products within the study area to identify those with the least potential environmental harm.
- identify key environmental hotspots in horticultural production.
- propose potential strategies to reduce the environmental burdens of horticultural products.

The function of the system is to produce fresh matured horticultural products at the farm gate. The functional unit for the analysis is defined as 1 t of freshly harvested horticultural product as declared by the grower between 2020 – 2023.

The LCI modeling follows an attributional framework, focusing on how the production of horticultural products influences environmental impacts and does not evaluate the consequences of horticultural product consumption [46]. Thus, the decision context for this study aligns with Situation A (micro-level decision support) as defined in the ILCD Handbook [40].The results are intended to inform decision-making at the farm level and within the horticultural supply chain without necessitating structural changes in the horticultural supply chain. The findings are aimed at a target audience comprising farmers, horticultural processors, agricultural experts, and policymakers to support strategies to reduce the environmental impact of the frozen vegetable supply chain.

The system boundary covers the cropping stage and follows a "cradle-to-farm gate" approach. The main activities included soil management, irrigation, fertilizer use, weed and pest control, and harvesting (Figure 4.1). All related direct and indirect emissions from the agrochemical use were also considered. No losses were assumed at this stage. Thus, no allocation was performed. Residual crop materials were also excluded due to unavailability of data.

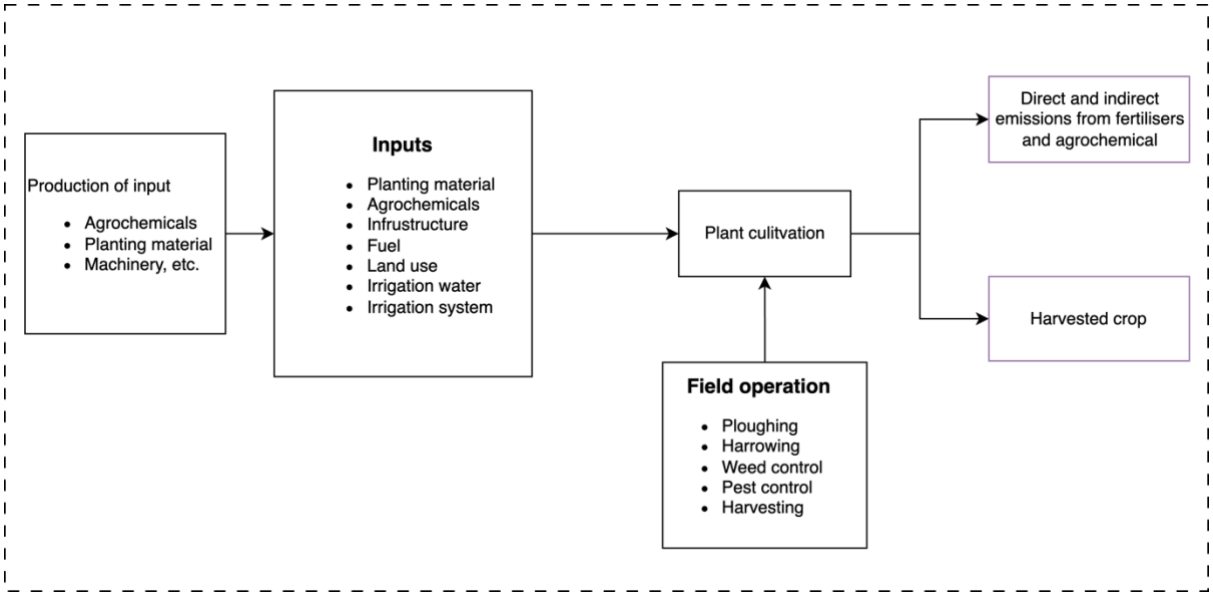



**Figure 4.1** The system boundaries of the cropping stage of the selected crops.

4.3.1.1 Data quality requirement

For this study, primary data was obtained from official farm documents from producers in the central and southern regions of Italy, namely Emilia-Romagna, Marche, Lazio, Umbria, Puglia, Molise, and Basilicata (illustrated in Figure 4.2). This data focuses on the cultivation and harvesting stages of the frozen vegetable production process. For a comprehensive analysis, background data

for upstream processes, such as farm inputs, ancillary materials, and machinery production, were included from the Ecoinvent version 3.9 database [168]. Ecoinvent is well-suited for analyzing activities within the food supply chain due to its extensive coverage of agricultural activities.

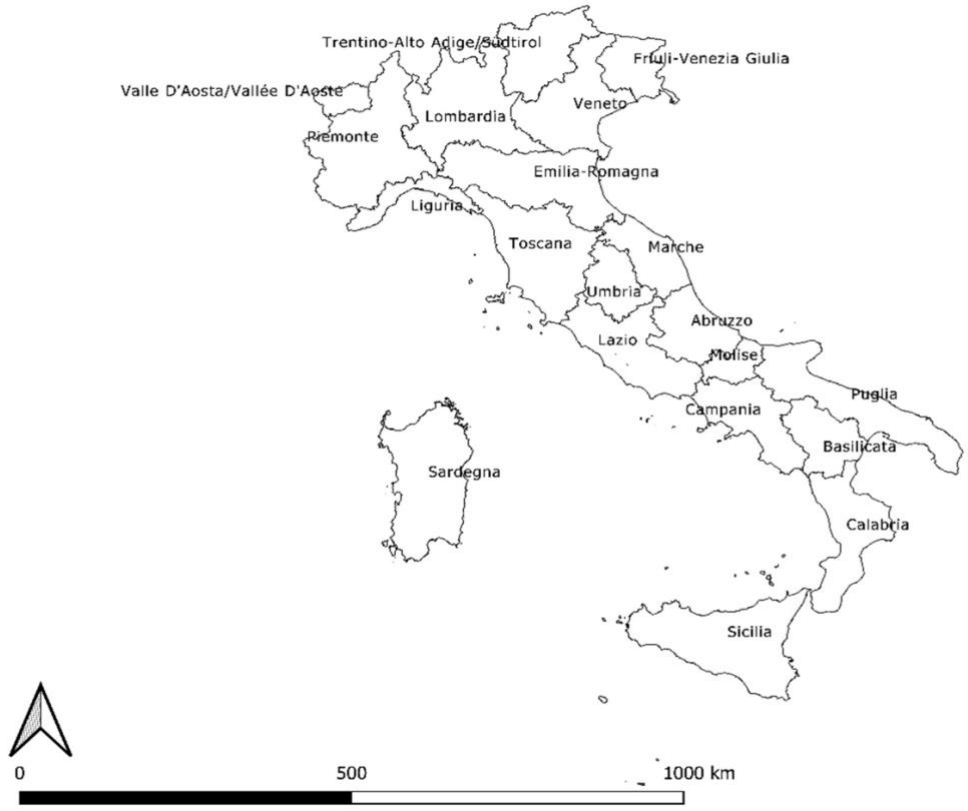



**Figure 4.2.** Regional map of Italy.

For conventional farms using fungicides, insecticides, and herbicides, it was assumed that 85% of the applied quantity entered the soil, 10% was released into the air, and 5% into water [169]. Organic farming excluded synthetic pesticide use. Due to the unavailability of region-specific data, the emissions calculation for fertilizer application relied on information from the Product Category Rules (PCR) for Arable and Vegetable Crops 2020 [170]. Emission factors for $NH_3$, NO, and $N_2O$–direct and indirect emissions in air, and nitrates and phosphorus from inorganic fertilizers and digestates emission in water were considered [171]. Bio-stimulants such as Spinosad, Kendal Te, and Impulsive Premium used in organic production were considered to have negligible impacts due to their natural production and lack of characterization factors. Indirect emissions from fuel combustion by farm machinery were included. However, this analysis did not account for nitrogen emissions from decomposing residues that were returned to the soil.

*4.3.2 Life cycle inventory*

4.3.2.1 Spinach

Spinach (*Spinacia oleracea* L.) is a dark green leafy vegetable of high economic and health importance. It is a rich source of many health-promoting compounds such as minerals, vitamins, phytochemicals, bioactive compounds, and fiber [172]. Due to the high abundance of phenolic compounds such as patuletin, spinacetin, glucuronides, and flavonoids, it is considered to have a comparatively high antioxidant capacity among vegetables [173–175]. Spinach extracts also have various functional properties, including antiproliferative, anti-obesity, hypoglycaemic, lipid-lowering, anti-inflammatory, and antioxidant [172, 175–178].

4.3.2.1.1 System description and LCI modeling framework

Spinach cultivation is conducted using standard agricultural practices in central Italy, where the climatic conditions range from humid subtropical to cold semi-arid. Spinach is typically grown in open fields, with two main sowing periods: November – February and July – September. The study included spinach farms varying in size from 1 to 27 ha, with an average of 6.43 ha. The yield per hectare ranged between 16.6 t/ha to 3.18 t/ha, with an average of 9.9 t/ha, due to factors such as soil type, climate conditions, and cultivar selection. Farmers practiced crop rotation, alternating spinach with cereals, other vegetables, and legumes to ensure soil health and optimize nutrient availability.

Appropriate land preparation activities, such as plowing and harrowing, were carried out to prepare the seedbed. The study considered organic and integrated farming systems which differ regarding input use, particularly fertilizers and pesticides. Organic farming relies solely on organic inputs, while integrated farming may incorporate limited use of synthetic inputs. Weed control is crucial for spinach production, as weeds compete with the crop for essential resources. Integrated farming systems typically rely on a combination of chemical herbicides, such as S-metolachlor, Metamitron, Lenacil, Phenmedipham, Quizalofop-p-ethyl, Propaquizafop, and Cycloxydim, applied before sowing and after emergence. On the other hand, organic farming employs cultural practices like crop rotation, intercropping, and hand weeding to suppress weed growth. Fertilization practices vary between the two systems. Integrated farming involves organic and synthetic fertilizers, including compost, manure, and mineral fertilizers. Organic agriculture relies solely on organic inputs, such as compost and manure, to improve soil fertility and provide essential nutrients to the crop. Irrigation is critical for spinach cultivation, especially during periods of low rainfall. The crop typically requires 25 – 38 $m^3$ of water/ha/ week. Sprinkler and furrow irrigation systems are

commonly used to deliver water to the crop. Spinach is harvested when the plants reach maturity, typically 2 – 3 months after sowing. The first harvest occurs when the plants have 5 – 6 leaves, and subsequent harvests can be retaken between 10 – 14 days, although the yield and quality of successive harvests may decrease. The average yield for the first cutting was 12,271 kg/ha, while the second was 11,982 kg/ha for harvested spinach.

This study used primary data collected from 73 farms in central and southern Italy, focusing on the 2019 and 2020 growing seasons. The farms were categorized based on geographic location and farming system (organic or integrated). Due to the absence of significant by-products or co-products, allocation was unnecessary. Primary data was collected for the cultivation and harvesting phases, while secondary data from the Ecoinvent database (v.3.9) was used for background processes. The inventory table is reported in Table 4.1.

**Table 4.1.** Main input and output flows of cultivated spinach (Average values/t fresh spinach).

| Process/Material | Unit | Integrated | Organic |
|---|---|---|---|
| Land | ha | 2.60E-01 | 2.54E-01 |
| Water for irrigation | $m^3$ | 2.24E+00 | 1.19E-01 |
| Water for agrochemical treatment | L | 1.38E+03 | 7.83E+02 |
| Seed | kg | 2.00E-01 | 2.00E-01 |
| Sowing | ha | 2.60E-01 | 2.54E-01 |
| Rolling | ha | 2.60E-01 | 2.54E-01 |
| Ploughing | ha | 2.60E-01 | 2.54E-01 |
| Rotary harrow | ha | 9.91E-02 | 2.54E-01 |
| Harrowing | ha | 1.38E-01 | 2.54E-01 |
| Weeding | ha | 1.31E-01 | 2.54E-01 |
| Milling | ha | 8.07E-03 | 2.54E-01 |
| Fertilising | ha | 6.37E-01 | 2.54E-01 |
| Application of plant protection | ha | 9.81E-01 | 1.36E+00 |
| Harvesting | ha | 1.27E-01 | 2.54E-01 |
| Vertical cutting | ha | 1.49E-01 | 2.54E-01 |
| Solid manure spreading | kg | 6.03E+02 | 6.03E+02 |
| Liquid Manure spreading | $m^3$ | 6.28E-01 | 6.28E-01 |
| Ammonium sulfate | kg | 7.06E-01 | – |
| Ammonium nitrate | kg | 2.94E-01 | – |
| Calcium nitrate | kg | 3.13E-01 | – |
| Urea | kg | 3.61E+01 | – |
| Diammonium phosphate | kg | 3.57E+01 | – |
| Monoammonium phosphate | kg | 3.23E-01 | – |
| Single superphostate | kg | 6.78E-01 | – |
| Inorganic K as $K_2O$ | kg | 8.46E-01 | – |
| Inorganic P as $P_2O_5$ | kg | 6.23E-01 | – |
| Organic N fertiliser | kg | 7.18E+00 | 7.18E+00 |
| Organic P fertiliser | kg | 9.66E-01 | 9.66E-01 |
| S-metolachlor | kg | 4.45E-02 | – |
| Metamitron | kg | 4.06E-01 | – |

| | | | |
|---|---|---|---|
| Lenacil | kg | 1.09E-01 | – |
| Phenmedipham | kg | 3.44E-02 | – |
| Quizalofop-p-ethyl | kg | 2.23E-02 | – |
| Propaquizafop | kg | 6.44E-02 | – |
| Cycloxydim | kg | 1.29E-02 | – |
| Etofenprox | kg | 5.08E-02 | – |
| Indoxacarb | kg | 2.35E-03 | – |
| Chlorantraniliprole | kg | 1.81E-02 | – |
| Iron phosphate | kg | 6.61E-01 | – |
| Cymoxanil (pure) | kg | 8.08E-03 | |
| Copper oxychloride | kg | 4.42E-02 | 3.83E-01 |
| Dimethomorph | kg | 3.32E-02 | – |
| Pyraclostrobin | kg | 1.86E-02 | – |
| Propamocarb | kg | 1.53E-02 | – |
| Boscalid | kg | 2.30E-02 | – |
| Sulfur | kg | – | 1.28 |
| Spinosad | kg | – | 3.75E-01 |
| *Emissions to air* | | | |
| $NH_3$-N | kg | 9.9E+00 | 1.51E+00 |
| $N_2O$-N (direct) | kg | 3.06E-01 | 1.58E-01 |
| $N_2O$-N (indirect) | kg | 3.06E-01 | 1.58E-01 |
| NO-N | kg | 8.56E-02 | 3.59E-02 |
| S-metolachlor | kg | 4.45E-03 | – |
| Metamitron | kg | 4.06E-02 | – |
| Lenacil | kg | 1.09E-02 | – |
| Phenmedipham | kg | 3.44E-03 | – |
| Quizalofop-p-ethyl | kg | 2.23E-03 | – |
| Propaquizafop | kg | 6.44E-03 | – |
| Cycloxydim | kg | 1.29E-03 | – |
| Etofenprox | kg | 5.08E-03 | – |
| Indoxacarb | kg | 2.35E-04 | – |
| Chlorantraniliprole | kg | 1.81E-03 | – |
| Iron phosphate | kg | 6.61E-02 | – |
| Cymoxanil (pure) | kg | 8.08E-04 | – |
| Copper oxychloride | kg | 4.42E-03 | 3.83E-02 |
| Dimetomorph | kg | 3.32E-03 | – |
| Pyraclostrobin | kg | 1.86E-03 | – |
| Propamocarb | kg | 1.53E-03 | – |
| Boscalid | kg | 2.30E-03 | – |
| Sulfur | kg | – | 0.128 |
| Spinosad | kg | – | 3.75E-02 |
| *Emissions to water* | | | |
| Nitrate (indirect) | kg | 3.34 | 1.72 |
| Phosphorus | kg | 0.898 | 4.83E-02 |
| S-metolachlor | kg | 2.23E-03 | – |
| Metamitron | kg | 2.03E-02 | – |
| Lenacil | kg | 5.45E-03 | – |
| Phenmedipham | kg | 1.72E-03 | – |
| Quizalofop-p-ethyl | kg | 1.12E-03 | – |
| Propaquizafop | kg | 3.22E-03 | – |
| Cycloxydim | kg | 6.45E-04 | – |

| | | | |
|---|---|---|---|
| Etofenprox | kg | 2.54E-03 | – |
| Indoxacarb | kg | 1.18E-04 | – |
| Chlorantraniliprole | kg | 9.05E-04 | – |
| Iron phosphate | kg | 3.31E-02 | – |
| Cymoxanil (pure) | kg | 4.04E-04 | – |
| Copper oxychloride | kg | 2.21E-03 | 1.92E-02 |
| Dimetomorph | kg | 1.66E-03 | |
| Pyraclostrobin | kg | 9.28E-04 | |
| Propamocarb | kg | 7.65E-04 | |
| Boscalid | kg | 1.15E-03 | |
| Sulfur | kg | – | 0.064 |
| Spinosad | kg | – | 1.88E-02 |
| *Emissions to soil* | | | |
| S-metolachlor | kg | 3.78E-02 | – |
| Metamitron | kg | 3.45E-01 | – |
| Lenacil | kg | 9.27E-02 | – |
| Phenmedipham | kg | 2.92E-02 | – |
| Quizalofop-p-ethyl | kg | 1.90E-02 | – |
| Propaquizafop | kg | 5.47E-02 | – |
| Cycloxydim | kg | 1.10E-02 | – |
| Etofenprox | kg | 4.32E-02 | – |
| Indoxacarb | kg | 2.00E-03 | – |
| Chlorantraniliprole | kg | 1.54E-02 | – |
| Iron phosphate | kg | 5.62E-01 | – |
| Cymoxanil (pure) | kg | 6.87E-03 | – |
| Copper oxychloride | kg | 3.75E-02 | 3.26E-01 |
| Dimetomorph | kg | 2.83E-02 | – |
| Pyraclostrobin | kg | 1.58E-02 | – |
| Propamocarb | kg | 1.30E-02 | – |
| Boscalid | kg | 1.96E-02 | – |
| Sulfur | kg | – | 1.09 |
| Spinosad | kg | – | 3.19E-01 |

#### 4.3.2.2 Green peas

Green peas (*Pisum sativum* L.) are a valuable legume crop, rich in essential nutrients such as protein, fiber, vitamins, and minerals [179]. The high fiber content contributes to digestive health and slows down starch digestion [180, 181]. The intermediate amylose content of pea starch further helps to reduce the glycaemic index of pea-based foods [182, 183]. Pea protein has been shown to have various health benefits, including potential antihypertensive and antioxidant properties [184, 185]. Additionally, the vitamins and minerals present in peas can help prevent nutrient deficiencies. Peas also contain phytochemicals, such as polyphenols, saponins, and oligosaccharides, linked to various health benefits, including antioxidant, anti-inflammatory, and prebiotic effects [179, 186–188].

4.3.2.2.1 System description and LCI modeling framework

Primary data were extracted from official farm documents sourced from six Italian regions: Marche, Emilia-Romagna, Puglia, Umbria, Molise, and Tuscany. A representative sample of 187 of 457 fields was analyzed, including 177 conventional and 10 organic fields, reflecting the regional production ratio. Cultivation occurred on open fields ranging from 2.5 to 28 ha, adhering to standard agricultural practices. Farmers implemented various field operations under organic and conventional systems, such as plowing, harrowing, sowing, and agrochemical applications. Planting occurred between November and January, with harvesting occurring from April to June, depending on the pea variety. Seed sowing rates averaged 200 – 250 kg/ha. Water usage for dissolving and diluting agrochemicals was calculated based on product label recommendations. Fertilizer and pesticide data were sourced directly from farmer records. Digestate quantities were estimated as averages of minimum and maximum application rates per hectare, considering nitrate-vulnerable zone regulations. Digestate composition was referenced from Möller and Müller [189]. Table 4.2 provides a detailed inventory of input data.

**Table 4.2.** Main input and output flows of cultivated peas (Average values/t of harvested peas).

| Input/Output | Unit | Integrated peas | Organic peas |
|---|---|---|---|
| Land surface | ha | 1.28E-01 | 1.24E-01 |
| Seeding rate | kg/ha | 5.12E+01 | 4.96E+01 |
| Harrowing | ha | 2.56E-01 | 2.48E-01 |
| Ploughing | ha | 2.56E-01 | 2.48E-01 |
| Sowing | ha | 2.56E-01 | 2.48E-01 |
| Hoeing | ha | 2.56E-01 | 2.48E-01 |
| Fertilising | ha | 2.56E-01 | 2.48E-01 |
| Application of plant protection | ha | 2.56E-01 | 2.48E-01 |
| Combine harvesting | ha | 2.56E-01 | 2.48E-01 |
| Solid manure spreading | kg | 3.66E+03 | 2.70E+03 |
| Liquid manure spreading | m3 | 3.81 | 3.68 |
| Ammonium nitrate | kg | 7.00E-01 | – |
| Calcium nitrate | kg | 2.50E-01 | – |
| Inorganic K as $K_2O$ | kg | 4.00E-02 | – |
| Inorganic P as $P_2O_5$ | kg | 9.60E-01 | – |
| Sodium borates | kg | 2.12E-04 | – |
| Molybdenum trioxide | kg | 1.15E-03 | – |
| Copper sulfate | kg | 1.30E-01 | – |
| Zinc oxide | kg | 3.30E-03 | – |
| sulfonly-urea compound | kg | 5.01E-02 | – |
| pyrethroid compound | kg | 2.50E-03 | – |
| Thio-carbamate compound | kg | 9.20E-02 | – |
| Pendimethalin | kg | 1.20E+01 | – |
| Aclonifen | kg | 1.40E-01 | – |
| Quicklime | kg | 4.44E-02 | – |
| Benzothiadazole | kg | 5.27E-02 | – |
| Phenoxy compound | kg | 5.23E-03 | – |

| | | | |
|---|---|---|---|
| Spinosad | kg | – | 1.00E-03 |
| Mineral oil | kg | – | 3.40E-02 |
| Gylphosate | kg | 7.63E-04 | – |
| Organic N fertiliser | kg | 4.35E+01 | 4.22E+01 |
| Organic P fertiliser | kg | 5.86 | 5.67 |
| *Emissions to air* | | | |
| $NH_3$-N | kg | 9.88 | 8.85 |
| $N_2O$-N (direct) | kg | 9.64E-01 | 9.28E-01 |
| $N_2O$-N (indirect) | kg | 9.64E-01 | 9.28E-01 |
| NO-N | kg | 2.42E-01 | 2.11E-01 |
| Sodium borates | kg | 7.00E-06 | – |
| Molybdenum trioxide | kg | 1.15E-04 | – |
| Copper sulfate | kg | 1.30E-02 | – |
| Zinc oxide | kg | 3.30E-04 | – |
| sulfonly-urea compound | kg | 5.01E-03 | – |
| pyrethroid compound | kg | 2.50E-04 | – |
| Thio-carbamate compound | kg | 9.20E-03 | – |
| Pendimethalin | kg | 1.20E+00 | – |
| Aclonifen | kg | 1.40E-02 | – |
| Quicklime | kg | 3.15E-03 | – |
| Benzothiadazole | kg | 5.27E-03 | – |
| Phenoxy compound | kg | 5.23E-04 | – |
| Gylphosate | kg | 7.63E-05 | – |
| Spinosad | kg | – | 1.00E-04 |
| Mineral oil | kg | – | 3.40E-03 |
| *Emissions to water* | | | |
| Nitrate (indirect) | kg | 1.05E+01 | 1.01E+01 |
| Phosphorus | kg | 3.14E-01 | 2.84E-01 |
| Sodium borates | kg | 3.50E-06 | – |
| Molybdenum trioxide | kg | 5.75E-05 | – |
| Copper sulfate | kg | 6.50E-03 | – |
| Zinc oxide | kg | 1.65E-04 | – |
| sulfonly-urea compound | kg | 2.51E-03 | – |
| pyrethroid compound | kg | 1.25E-04 | – |
| thio-carbamate compound | kg | 4.60E-03 | – |
| Pendimethalin | kg | 6.00E-01 | – |
| Aclonifen | kg | 7.00E-03 | – |
| Quicklime | kg | 1.58E-03 | – |
| Benzothiadazole | kg | 2.64E-03 | – |
| Phenoxy compound | kg | 2.62E-04 | – |
| Gylphosate | kg | 3.82E-05 | – |
| Spinosad | kg | – | 5.00E-05 |
| Mineral oil | kg | – | 1.70E-03 |
| *Emissions to soil* | | | |
| Sodium borates | kg | 1.80E-04 | – |
| Molybdenum trioxide | kg | 9.78E-04 | – |
| Copper sulfate | kg | 1.11E-01 | – |
| Zinc oxide | kg | 2.81E-03 | – |
| sulfonly-urea compound | kg | 4.26E-02 | – |
| pyrethroid compound | kg | 2.13E-03 | – |
| thio-carbamate compound | kg | 7.82E-02 | – |

| | | | |
|---|---|---|---|
| Pendimethalin | kg | 1.02E+01 | – |
| Aclonifen | kg | 1.19E-01 | – |
| Quicklime | kg | 2.68E-02 | – |
| Benzothiadazole | kg | 4.48E-02 | – |
| Phenoxy compound | kg | 4.45E-03 | – |
| Gylphosate | kg | 6.49E-04 | – |
| Spinosad | kg | – | 8.50E-04 |
| Mineral oil | kg | – | 2.89E-02 |

#### 4.3.2.3 Onion

Onion (*Allium cepa* L.) is a popular vegetable in various colors, with widespread global consumption. As a nutrient-rich food, onions are enjoyed both fresh and dried and are a staple in many cultures worldwide [190]. Onions are rich sources of bioactive compounds, including flavonoids, phenolic compounds, and organo-sulfur compounds, which offer potential health benefits such as antioxidant, anti-inflammatory, antimicrobial, and anticancer properties [191, 192].

##### 4.3.2.3.1 System description and LCI modeling framework

Data was collected from 26 farms in the Puglia region, covering onion cultivated between 2020 and 2022. Fifteen of the farms were under an integrated system, while 11 of the farms had organic production. The cultivation process begins with land preparation, which includes tillage operations. Onions can be cultivated using seeds, transplants, or sets. When grown from seeds, onions are typically planted before winter (in November), while cultivation using transplants occurs in spring, around April. This study used transplants, as this was the process available in the Ecoinvent database. Following planting, frequent irrigation is essential to support seed germination, leaf growth, bulb development, and sprouting, as these stages are highly dependent on soil moisture or environmental humidity. Fertilizers, primarily ammonium nitrate and diammonium phosphate, were applied during the cultivation process in the integrated system. However, no synthetic fertilizers were used in the organic system. Various plant protection compounds to manage pests and diseases, such as Downy mildew and powdery mildew were used. In the case of organic production, Copper sulfate, hydrated lime, sulfur, and essential oil were used for pest control. Harvesting typically occurs between June and August when onions are dried and their foliage is withered using mechanical onion harvesters. The average yield was 54 t/ha for integrated production and 42 t/ha for organic production. Table 4.3 summarizes the average inventory data for onion cultivation, detailing key inputs such as seeds, fertilizers, pesticides, water, and energy, along with outputs including direct and indirect emissions associated with agrochemical use.

**Table 4.3.** Main input and output flows of cultivated onion (Average values/t of harvested onion).

| Inputs/Outputs | Unit | Integrated | Organic |
|---|---|---|---|
| Land Use | ha | 1.84E-02 | 2.38E-02 |
| Irrigation | $m^3$ | 1.52E+01 | 2.22E+01 |
| Ploughing | ha | 1.84E-02 | 2.38E-02 |
| Harrowing | ha | 1.84E-02 | 2.38E-02 |
| Harrowing | ha | 1.84E-02 | 2.38E-02 |
| Rolling | ha | 1.84E-02 | 2.38E-02 |
| Sowing | ha | 1.84E-02 | 2.38E-02 |
| Plant protection application | ha | 1.84E-02 | 2.38E-02 |
| Fertilizing | ha | 1.84E-02 | 2.38E-02 |
| Harvesting | ha | 1.84E-02 | 2.38E-02 |
| Planting material | p | 2.05E+04 | 2.05E+04 |
| *Fertilizer* | | | |
| Ammonium nitrate -$NH_4NO_3$ | kg | 1.90 | – |
| Diammonium phosphate | kg | 3.10 | – |
| *Plant protection* | | | |
| Copper | kg | 1.96E-01 | 1.80E-01 |
| Sulfur | kg | 1.76E-01 | 1.36E-01 |
| Aclonifen | kg | 1.34E-03 | – |
| Clopiralid | kg | 1.19E-03 | – |
| Copper-Oxychloride | kg | 9.31E-03 | – |
| Azoxystrobin | kg | 2.48E-03 | – |
| Spinosad | kg | – | 4.13E-03 |
| Azadiractina | kg | – | 1.68E-03 |
| Essential oil | kg | – | 5.16E-03 |
| Copper-Sulphate | kg | – | 3.27E-02 |
| Pesticide (unspecified) | kg | 9.45E-01 | 3.93E-01 |
| Herbicide (unspecified) | kg | 8.02E-03 | – |
| *Emissions to air* | | | |
| $NH_3$-N | kg | 1.93E-02 | – |
| $N_2O$-N | kg | 6.76E-03 | – |
| NO-N | kg | 5.28E-03 | – |
| Copper | kg | 1.31E-03 | 2.56E-03 |
| Aclonifen | kg | 3.93E-04 | – |
| Bromoxynil | kg | 1.56E-04 | – |
| Sulfur | kg | 5.07E-02 | 9.31E-03 |
| Copper-Oxychloride | kg | 8.38E-04 | – |
| Azoxystrobin | kg | 2.24E-04 | – |
| Spinosad | kg | – | 3.71E-04 |
| Azadiractina | kg | – | 1.51E-04 |
| Essential oil | kg | – | 4.64E-04 |
| Copper-Sulphate | kg | – | 2.94E-03 |
| *Emissions to water* | | | |
| Nitrate (indirect) | kg | 1.04E-01 | – |
| Phosphorus | kg | 1.25E-02 | – |
| Copper | kg | 1.46E-04 | 2.84E-04 |
| Aclonifen | kg | 4.37E-05 | – |
| Bromoxynil | kg | 1.74E-05 | – |
| Sulfur | kg | 5.63E-03 | 8.46E-04 |
| Copper-Oxychloride | kg | 9.31E-05 | – |

| | | | |
|---|---|---|---|
| Azoxystrobin | kg | 2.48E-05 | – |
| Spinosad | kg | – | 4.13E-05 |
| Azadiractina | kg | – | 1.68E-05 |
| Essential oil | kg | – | 5.16E-05 |
| Copper-Sulphate | kg | – | 3.27E-04 |
| *Emissions to soil* | | | |
| Copper | kg | 2.28E-02 | 2.56E-02 |
| Aclonifen | kg | 3.93E-03 | – |
| Bromoxynil | kg | 1.56E-03 | – |
| Sulfur | kg | 4.75E-01 | 7.61E-02 |
| Copper-Oxychloride | kg | 8.38E-03 | – |
| Azoxystrobin | kg | 2.24E-03 | – |
| Spinosad | kg | – | 3.71E-03 |
| Azadiractina | kg | – | 1.51E-03 |
| Essential oil | kg | – | 4.64E-03 |
| Copper-Sulphate | kg | – | 2.94E-02 |
| *Indirect emissions* | | | |
| $N_2O$ | kg | 1.01E-02 | – |

#### 4.3.2.4 Tomato

Tomato (*Lycopersicon esculentum*) is a significant Solanaceae family member, widely cultivated as a food and cash crop. It is marketed fresh or processed into various forms like paste, whole, or cut, and can also be frozen or canned [193]. Tomatoes are a rich source of essential nutrients and phytochemicals, notably lycopene, a carotenoid linked to potential health benefits. The abundance of carotenoids that confer antioxidant, anti-inflammatory, and anticancer properties contributes to preventing degenerative diseases like cancer and cardiovascular disease [194].

##### 4.3.2.4.1 System description and LCI modeling framework

Data was collected from 116 tomato farms of varying sizes, covering a total cultivated area of approximately 509 ha and producing an estimated 9,083.51 t of fresh tomatoes during the 2020 production year. The average yield was 40 t/ha. Most fields were in Puglia (112 fields), with a few in Tuscany (3) and Emilia-Romagna (1). All farms employed integrated cultivation practices. Pre-planting field operations included tillage, plowing, and harrowing. Tomato seedlings were transplanted between April and May. Subsequent cultural practices involved irrigation and fertilization supported plant growth. Fertilizers were applied to the soil or the plants, while herbicides were used to control weeds competing for nutrients. Fungicides and insecticides were applied to manage fungal infections, such as *Phytophthora infestans*, *Alternaria solani, Leveilulla taurica*, and powdery mildew—and insect pests like *Spodoptera littoralis*, *Tuta absoluta*, and *Heliothis armigera*. Harvesting occurred between July and August. The main inputs and outputs of cultivated tomatoes are reported in Table 4.4.

**Table 4.4**. Main input and output flows of cultivated tomato (Average values/t harvested tomato).

| Input/Output | Unit | Amount/FU |
|---|---|---|
| Water, well, IT | $m^3$ | 1.55E+01 |
| Occupation, arable land, unspecific use | ha a | 1.82E-02 |
| Water, process, unspecified natural origin/m3 | $m^3$ | 6.45E-01 |
| Water, process, unspecified natural origin/m3 | $m^3$ | 1.96E+00 |
| Tillage, ploughing | ha | 1.82E-02 |
| Tillage, curry, by weeder | ha | 1.92E-02 |
| Tillage, harrowing, by rotary harrow | ha | 1.82E-02 |
| Fertilizing | ha | 1.82E-02 |
| Application of plant protection | ha | 1.82E-02 |
| Harvesting, by complete harvester, beets | ha | 1.82E-02 |
| NPK 15-15-15 | kg | 6.16E+00 |
| Diammonium phosphate (18%N, 46% p2o5) | kg | 3.18E+00 |
| Ammonium sulfate | kg | 1.05E+00 |
| Urea (46% N) | kg | 8.26E-01 |
| Calcium nitrate | kg | 1.14E+00 |
| Dithiocarbamate-compound | kg | 4.14E-03 |
| Copper sulfate | kg | 1.34E-02 |
| Pesticide unspecficied | kg | 5.16E-01 |
| *Emissions to air* | | |
| Ammonia | kg | 2.23E-01 |
| Nitrogen monoxide | kg | 5.47E-02 |
| Dinitrogen monoxide | kg | 2.67E-02 |
| Dinitrogen monoxide (indirect emission) | kg | 3.02E-02 |
| Copper oxychloride | kg | 9.52E-03 |
| Metiram | kg | 1.11E-03 |
| Pyraclostrobin | kg | 2.86E-03 |
| Copper sulfate | kg | 1.40E-02 |
| Tefluthrin | kg | 1.67E-02 |
| Ketones | kg | 2.37E-03 |
| Abamectin | kg | 1.13E-03 |
| Cypermethrin | kg | 7.58E-04 |
| Indoxacarb | kg | 4.05E-04 |
| Lambda-cyhalothrin | kg | 9.36E-03 |
| Pendimethalin | kg | 2.95E-03 |
| Metolachlor, s | kg | 1.49E-03 |
| Metribuzin | kg | 5.55E-04 |
| Copper sulfate, tri-basic | kg | 1.59E-02 |
| Metalaxyl-M | kg | 1.04E-02 |
| Cymoxanil | kg | 2.16E-03 |
| Sulfur | kg | 9.03E-03 |
| Ryanodine | kg | 2.67E-04 |
| Rimsulfuron | kg | 7.28E-05 |
| Azoxystrobin | kg | 1.26E-03 |
| Propaquizafop | kg | 1.73E-03 |
| Deltamethrin | kg | 1.04E-03 |
| Formetanate hydrochloride | kg | 1.08E-03 |
| Dimethomorph | kg | 3.52E-03 |
| Acetamiprid | kg | 3.65E-03 |
| Emamectin benzoate | kg | 4.72E-03 |

| | | |
|---|---|---|
| Cycloxydim | kg | 2.12E-03 |
| Difenoconazole | kg | 1.50E-03 |
| Zoxamide | kg | 2.20E-03 |
| Roundup | kg | 8.06E-03 |
| Acibenzolar-s-methly | kg | 1.53E-04 |
| Propamocarb HCl | kg | 3.33E-03 |
| *Emissions to water* | | |
| Phosphate (groundwater) | kg | 6.65E-04 |
| Phosphate (river) | kg | 1.66E-03 |
| Phosphate (river) | kg | 1.11E-02 |
| Nitrate compounds | kg | 3.19E-02 |
| Copper oxychloride | kg | 5.13E-03 |
| Metiram | kg | 8.53E-04 |
| Pyraclostrobin | kg | 2.13E-03 |
| Copper sulfate | kg | 7.44E-03 |
| Tefluthrin | kg | 9.34E-03 |
| Insecticides, unspecified | kg | 1.66E-03 |
| Abamectin | kg | 5.76E-04 |
| Cypermethrin | kg | 3.86E-04 |
| Indoxacarb | kg | 2.21E-04 |
| Lambda-cyhalothrin | kg | 4.69E-03 |
| Pendimethalin | kg | 1.44E-03 |
| Metolachlor, s | kg | 8.50E-04 |
| Metribuzin | kg | 2.77E-04 |
| Copper sulfate, tri-basic | kg | 8.12E-03 |
| Metalaxyl-M | kg | 5.47E-03 |
| Cymoxanil | kg | 1.14E-03 |
| Sulfur | kg | 4.81E-03 |
| Ryanodine | kg | 1.43E-04 |
| Rimsulfuron | kg | 4.13E-05 |
| Azoxystrobin | kg | 7.97E-04 |
| Propaquizafop | kg | 1.09E-03 |
| Deltamethrin | kg | 5.36E-04 |
| Formetanate hydrochloride | kg | 5.43E-04 |
| Dimethomorph | kg | 1.76E-03 |
| Acetamiprid | kg | 1.89E-03 |
| Emamectin benzoate | kg | 2.56E-03 |
| Cycloxydim | kg | 1.06E-03 |
| Difenoconazole | kg | 7.51E-04 |
| Zoxamide | kg | 1.10E-03 |
| Roundup | kg | 1.73E-03 |
| Acibenzolar-s-methly | kg | 7.67E-05 |
| Propamocarb HCl | kg | 1.67E-03 |
| *Emissions to soil* | | |
| Copper oxychloride | kg | 8.71E-02 |
| Metiram | kg | 1.13E-02 |
| Pyraclostrobin | kg | 2.84E-02 |
| Copper sulfate | kg | 1.19E-01 |
| Tefluthrin | kg | 1.50E-01 |
| Insecticides, unspecified | kg | 3.16E-02 |
| Abamectin | kg | 1.08E-02 |

| | | |
|---|---|---|
| Cypermethrin | kg | 5.77E-03 |
| Indoxacarb | kg | 3.56E-03 |
| Lambda-cyhalothrin | kg | 7.77E-02 |
| Pendimethalin | kg | 2.82E-02 |
| Metolachlor, s | kg | 1.44E-02 |
| Metribuzin | kg | 4.61E-03 |
| Copper sulfate, tri-basic | kg | 1.50E-01 |
| Metalaxyl-M | kg | 9.30E-02 |
| Cymoxanil | kg | 1.84E-02 |
| Sulfur | kg | 8.11E-02 |
| Ryanodine | kg | 2.47E-03 |
| Rimsulfuron | kg | 7.10E-04 |
| Azoxystrobin | kg | 1.34E-02 |
| Propaquizafop | kg | 1.47E-02 |
| Deltamethrin | kg | 8.70E-03 |
| Formetanate hydrochloride | kg | 9.22E-03 |
| Dimethomorph | kg | 3.21E-02 |
| Acetamiprid | kg | 3.36E-02 |
| Emamectin benzoate | kg | 4.01E-02 |
| Cycloxydim | kg | 1.80E-02 |
| Difenoconazole | kg | 1.28E-02 |
| Zoxamide | kg | 1.87E-02 |
| Roundup | kg | 2.93E-02 |
| Acibenzolar-s-methly | kg | 1.30E-03 |
| Propamocarb HCl | kg | 2.83E-02 |

### *4.3.3 Interpretation*

The absolute midpoint characterization results for each cultivated product expressed per the selected functional unit for the two farming systems (integrated and organic) are presented. Results are reported across multiple impact categories. However, greater emphasis is placed on climate change due to its global importance. Furthermore, a detailed analysis of the contributing phases, key processes, and substances (hotspot analysis) is provided to interpret the findingsv.

## 4.4 RESULTS AND DISCUSSION

The cradle-to-farm gate midpoint impact category results of the assessed horticultural products are reported in Table 4.5. The average results are reported for integrated and organic production systems. From the results obtained, the organic products had a relatively better environmental performance than the integrated products due to the absence of synthetic agrochemicals and their related direct and indirect emissions. The environmental performance of the crops varied across the various impact categories. However, among the products assessed, tomato had the lowest

scores across all impact categories, except for WU. This was mainly due to the high yield output per hectare and low inputs.

*4.4.1 Onion*

Although organic onion production had lower environmental impacts across various categories, the differences between onions produced using integrated and organic systems were not substantial. The integrated system achieved a higher yield than the organic. However, its reliance on agrochemicals led to comparatively greater environmental impacts. Moreover, the absence of characterization factors for certain organic agrochemicals, such as essential oils, may have biased the results in favor of the organic system. Additionally, the integrated system showed greater variability due to the larger number of fields involved, making direct comparisons between the two systems challenging. As a result, it remains inconclusive which system is more sustainable.

From the relative contribution analysis of integrated onion production, seedling production was the primary contributor to most environmental impacts, except for LU and WU (Figure 4.3). In contrast, for organic onion production, the main contributors were emissions from agrochemicals, mainly copper sulfate, and field operations (Figure 4.4).

The CC scores were 92 kg $CO_2$ eq./t for integrated onions and 84 kg $CO_2$ eq./t for organic onions. These values align with previously reported cradle-to-farm gate CC scores for small-scale organic onion production, such as 79 kg $CO_2$ eq./t, where the main impacts were attributed to fuel use and fertilizer application [195]. However, Jensen et al.[196] reported a higher CC value for organic onions (139 kg $CO_2$ eq./t), with seedling production and $N_2O$ and $CO_2$ emissions being the largest contributors. Another estimate for organic onion production in the UK reported a CC score of 160 kg $CO_2$ eq./t [197], while another study cited even higher values, such as 210 kg $CO_2$ eq./t for UK-produced onions and 225 kg $CO_2$ eq./t for imported onions [198]. A high CC score of 324 kg $CO_2$ eq./t was also reported for onion production in Iran, with machinery for field operations identified as the key hotspot [198]. However, many of these studies did not provide an average yield per hectare, which limits further direct comparisons.

**Table 4.5.** Cradle-to-farm gate LCIA results for fresh horticultural products (expressed per tonne fresh weight of harvest product).

| Impact | Unit | Onion int. | Onion org. | Pea int. | Pea org. | Spinach int. | Spinach org. | Tomato int. |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 1.94 | 1.78 | 3.11 | 2.75 | 3.45 | 1.55 | 0.57 |
| CC | kg $CO_2$ eq. | 91.62 | 84.16 | 341.93 | 294.69 | 423.71 | 238.93 | 54.79 |
| ETF | CTUe | 1.43E+03 | 1.75E+03 | 1.20E+05 | 2.93E+03 | 8.85E+04 | 2.10E+03 | 2.25E+07 |
| PM | disease inc. | 1.55E-05 | 1.44E-05 | 2.17E-04 | 1.94E-04 | 2.22E-04 | 3.81E-05 | 7.15E-06 |
| MEU | kg N eq. | 0.42 | 0.41 | 4.62 | 4.43 | 2.29 | 1.05 | 0.13 |
| FEU | kg P eq. | 0.021 | 0.019 | 0.142 | 0.135 | 0.361 | 0.046 | 0.013 |
| TEU | mol N eq. | 7.50 | 7.41 | 90.72 | 81.35 | 89.62 | 17.98 | 3.14 |
| HT_c | CTUh | 7.10E-08 | 7.63E-08 | 1.72E-07 | 1.49E-07 | 1.94E-07 | 1.45E-07 | 2.66E-08 |
| HT_nc | CTUh | 2.14E-06 | 2.56E-06 | 6.92E-06 | 3.96E-06 | 8.19E-06 | 2.24E-06 | 6.88E-05 |
| IR | kBq U-235 eq. | 2.12 | 1.66 | 5.49 | 4.65 | 6.04 | 3.83 | 1.73 |
| LU | Pt | 1.68E+04 | 2.17E+04 | 1.17E+05 | 1.13E+05 | 2.22E+05 | 2.15E+05 | 1.55E+04 |
| ODP | kg CFC11 eq. | 1.79E-06 | 1.45E-06 | 6.47E-06 | 4.17E-06 | 4.34E-06 | 3.63E-05 | 9.27E-06 |
| POF | kg NMVOC eq. | 0.65 | 0.68 | 2.17 | 1.98 | 2.07 | 1.65 | 0.31 |
| FRD | MJ | 1.14E+03 | 1.09E+03 | 2.40E+03 | 2.17E+03 | 3.46E+03 | 1.98E+03 | 5.29E+02 |
| MRD | Kg Sb eq. | 1.09E-03 | 1.49E-03 | 1.85E-03 | 1.26E-03 | 4.91E-03 | 1.20E-03 | 4.13E-04 |
| WU | $m^3$ depriv. | 714.11 | 1022.98 | 232.31 | 222.36 | 410.49 | 99.40 | 830.14 |

Int – integrated production, org – Organic production. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

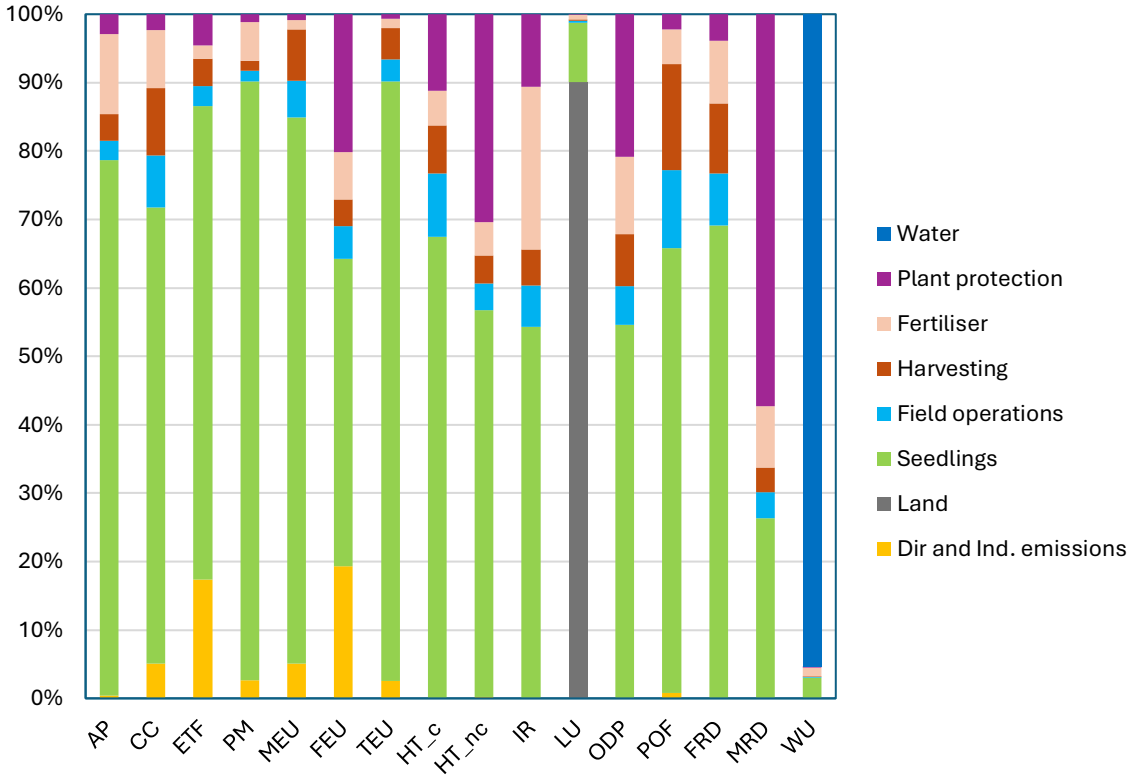



**Figure 4.3.** Relative contribution analysis (onion – integrated production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

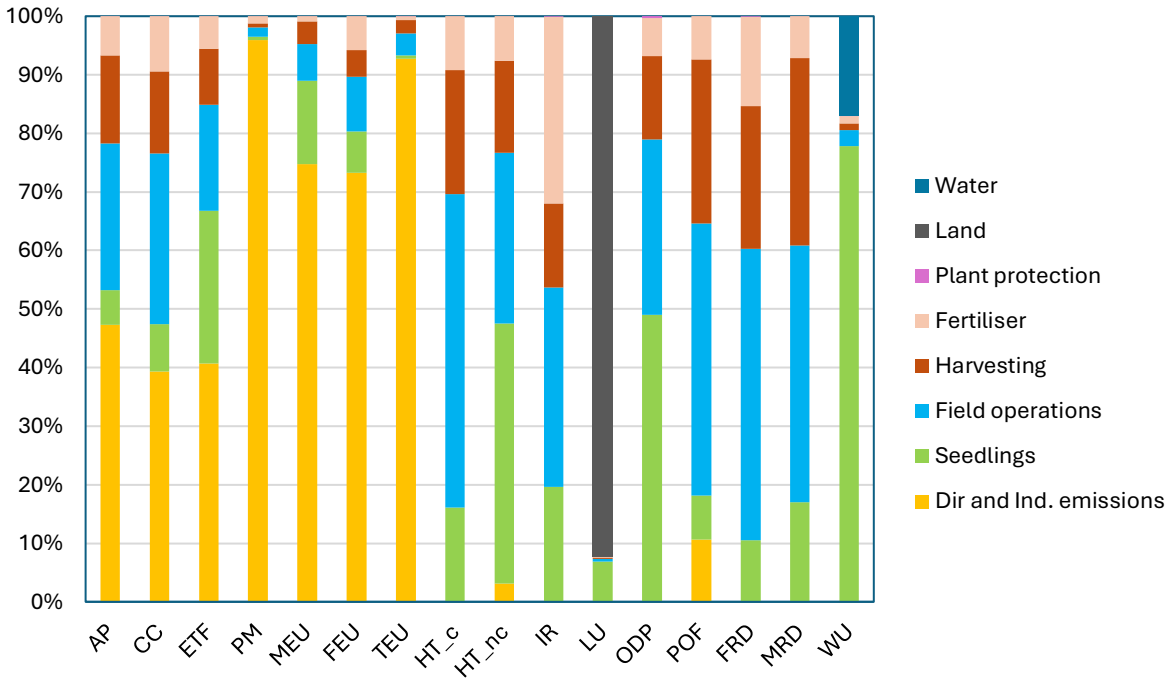



**Figure 4.4.** Relative contribution analysis (onion – organic production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### *4.4.2 Green pea*

The environmental impact results of integrated and organic pea cultivation suggest that integrated pea production has on average, a higher environmental impact than organic pea production (Table 4.5). In the context of organic farms, fewer synthetic chemicals are employed, and the use of biostimulants presents low environmental impacts. Moreover, plant protection products were excluded from impact calculation due to a lack of background processes for these compounds. It is also important to acknowledge that some organic farms recorded no yields and were consequently excluded from the dataset under consideration. Therefore, although organic farming practices may have a reduced impact due to the limited control available to farmers to address unexpected adversities like pest attacks swiftly, it is worth noting that significantly higher impacts may be reported

if minimal or no yields are obtained despite the resources expended, thus undermining environmental sustainability.

Furthermore, the allocation of land resources influences the environmental performance of both systems. Due to the restrictions on agrochemical use in organic production, farmers tend to prioritize using high-quality land with optimal soil conditions and favorable climates. However, for integrated farming systems, farmers can potentially use less suitable lands, relying on agrochemical inputs to compensate for soil limitations, weed pressure, and pest infestations. This disparity in land allocation strategies can significantly influence the observed environmental impacts and complicate direct comparisons between the two systems. In addition, it is important to mention that the analysis is based on a limited sample size of organic farms, 403 ha for organic farms and 4155.50 ha for integrated system. This disparity in the number of integrated and organic fields is a major limitation in comparing the environmental performance of the two systems. Therefore, a more comprehensive analysis with a larger and more representative sample of organic farms is necessary to draw definitive conclusions about the relative sustainability of organic and integrated spinach production systems.

The primary drivers of environmental impact varied across different impact categories. The most significant drivers for CC were direct emissions from agrochemicals, primarily $N_2O$, field operations, and input materials like pea seeds and ammonium nitrate. In integrated pea cultivation, agrochemicals for plant protection also made significant contributions to ODP and MRD. The relative contributions for integrated and organic pea production are illustrated in Figures 4.5 and 4.6, respectively.

The CC scores were 342 kg $CO_2$ eq./t and 295 kg $CO_2$ eq./t for integrated and organic peas, respectively. These results are comparable to those reported for field pea production in the USA (320 kg $CO_2$ eq./t) [199] but are higher than previous studies, which ranged from 130 to 240 kg $CO_2$ eq./t [200–202]. However, another study reported a higher CC score (485 kg $CO_2$ eq./t) for field peas production in Norway [203]. Differences in crop yields may contribute to the variations in results. For example, this study's average crop yield per hectare was 4 t, whereas Del Borghi et al. [201] reported a higher value of 5 to 5.5 t. Moreover, differences in data sources and uncertainties can influence the results, as some studies used secondary data [199, 200].

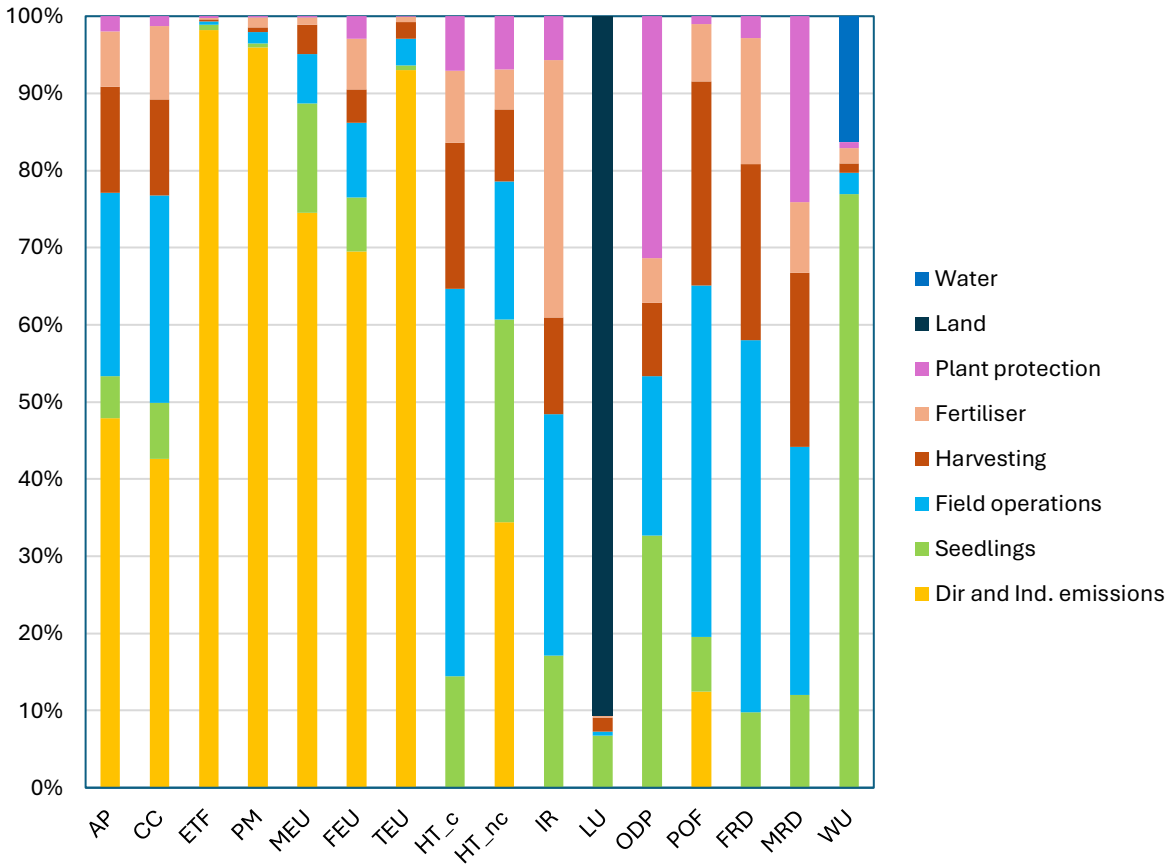



**Figure 4.5.** Relative contribution analysis (pea – integrated production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

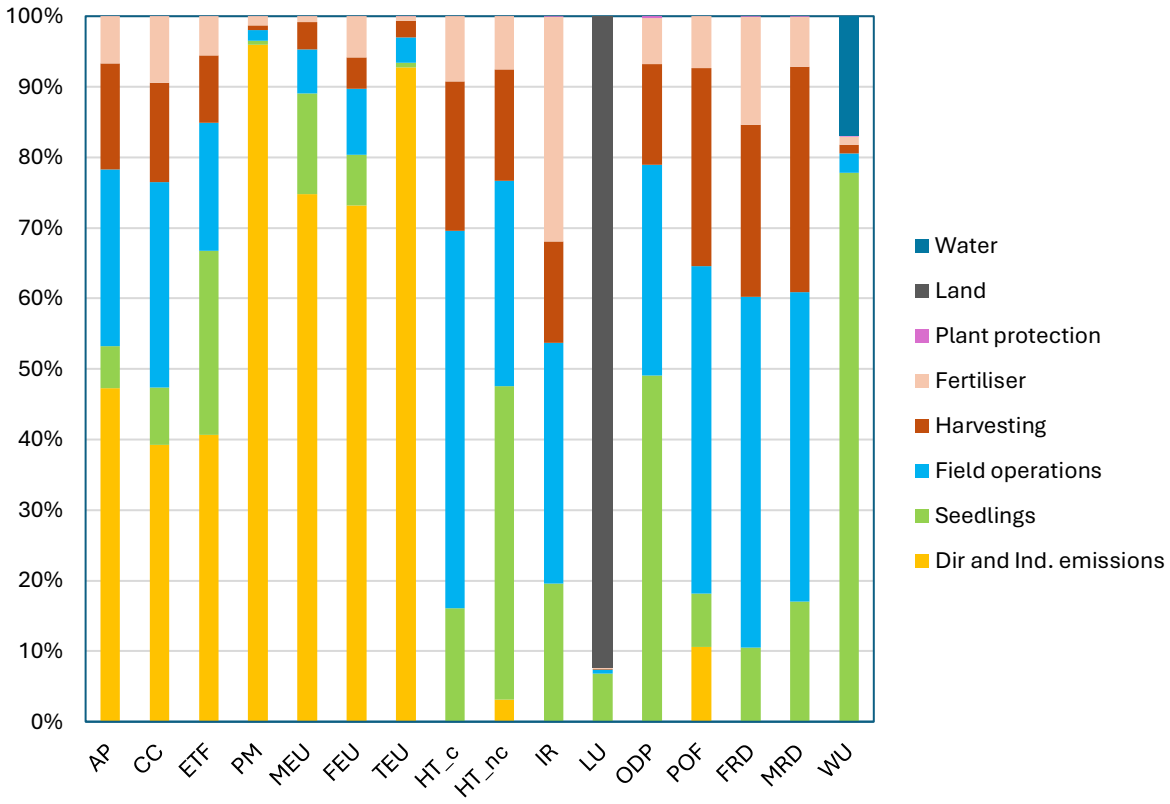



**Figure 4.6.** Relative contribution analysis (pea – organic production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### *4.4.3 Spinach*

The environmental impact of integrated spinach production was relatively higher than that of organic spinach across most impact categories (Table 4.5). However, this does not necessarily indicate that the organic system is more sustainable than the integrated system. A key limitation of this comparison is the disparity in sample sizes, as only three organic farms were assessed compared to seventy integrated farms, a discrepancy that reflects the current distribution of these farming systems. Consequently, the variability within the integrated system was significantly higher than within the organic system. The lower

environmental impacts observed in organic spinach production were largely attributed to its higher average yield (14.7 t/ha) compared to integrated farming (9.8 t/ha). However, when considering the standard deviation of integrated farm yields (±6.5 t/ha), some integrated farms performed better than organic farms, while others performed worse. Additionally, the reduced use of agrochemicals in organic farming contributed to its relatively lower environmental impact.

Furthermore, organic production is inherently more restrictive regarding agrochemical use, which means that only the most suitable fields, those with optimal soil quality and favorable climatic conditions, are typically allocated for organic cultivation. In contrast, integrated farming systems often include lower-quality fields, as the flexibility in agrochemical application allows farmers to compensate for soil fertility limitations, weed infestations, and pest pressures. This difference in land allocation strategy further complicates direct comparisons between the two systems. Thus, while the results suggest that organic spinach production may have lower environmental impacts in certain aspects, a more comprehensive analysis with a larger sample size of organic farms is necessary to draw definitive conclusions about the relative sustainability of these two production systems.

Regarding the contribution analysis, for the integrated production system, environmental impacts were primarily attributed to fertilizer and pesticide use (Figures 4.7 and 4.8). The highest impact shares for organic spinach were from field operations (38%) and emissions from digestate (37%). In integrated spinach production, direct emissions from nitrogen-based fertilizers contributed 35% to the total CC impact, with indirect emissions from fertilizer synthesis and use accounting for an additional 30%. However, it is worth noting that the sample size for organic farms was significantly smaller (10%) compared to integrated farms.

Regarding CC, the impact score for integrated spinach (424 kg $CO_2$ eq./t) was almost double that of organic spinach (239 kg $CO_2$ eq./t), primarily driven by agrochemical emissions. These results are consistent with the CC value of 230 kg $CO_2$ eq./t reported for spinach produced in a Japanese greenhouse system [204]. However, other studies reported lower values for organic spinach under polytunnel cultivation and (94 – 115 kg $CO_2$ eq./t for organic spinach under polytunnel cultivation) [205], 170 kg $CO_2$ eq./t was

reported for spinach cultivation in the UK [200]. These differences may be attributed to factors such as cultivation system, farm size, climatic conditions, modeling choices, and the inclusion of other phases like packaging within the system boundary.

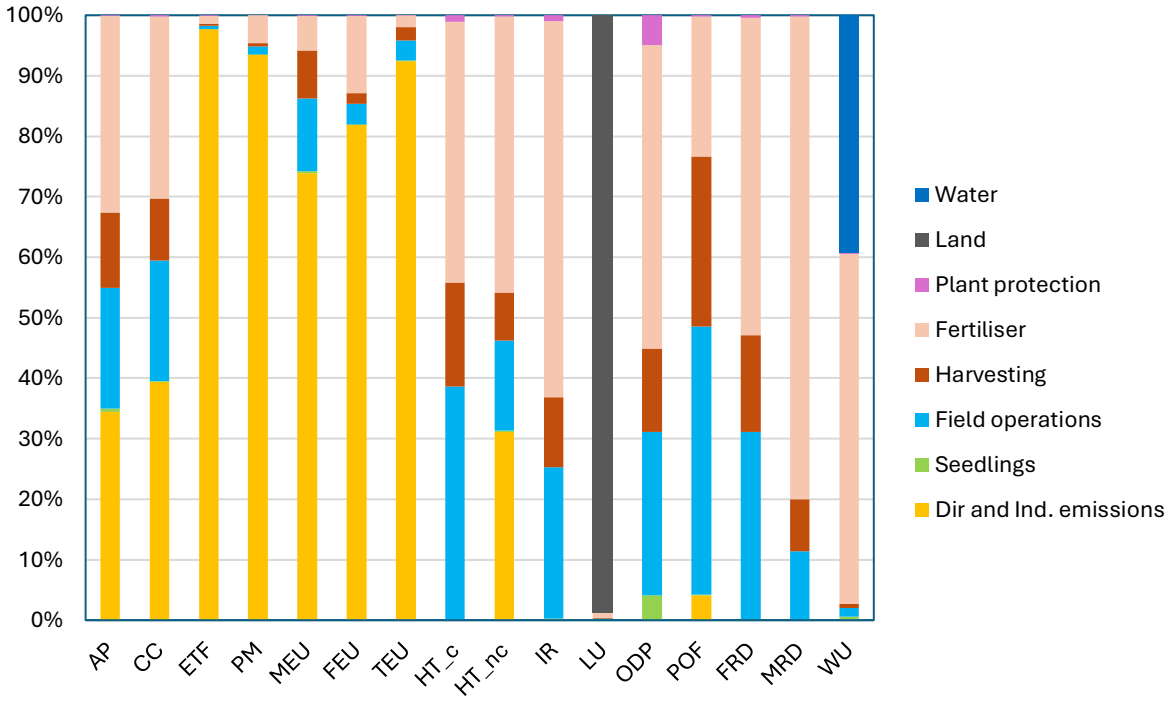



**Figure 4.7.** Relative contribution analysis (spinach – integrated production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

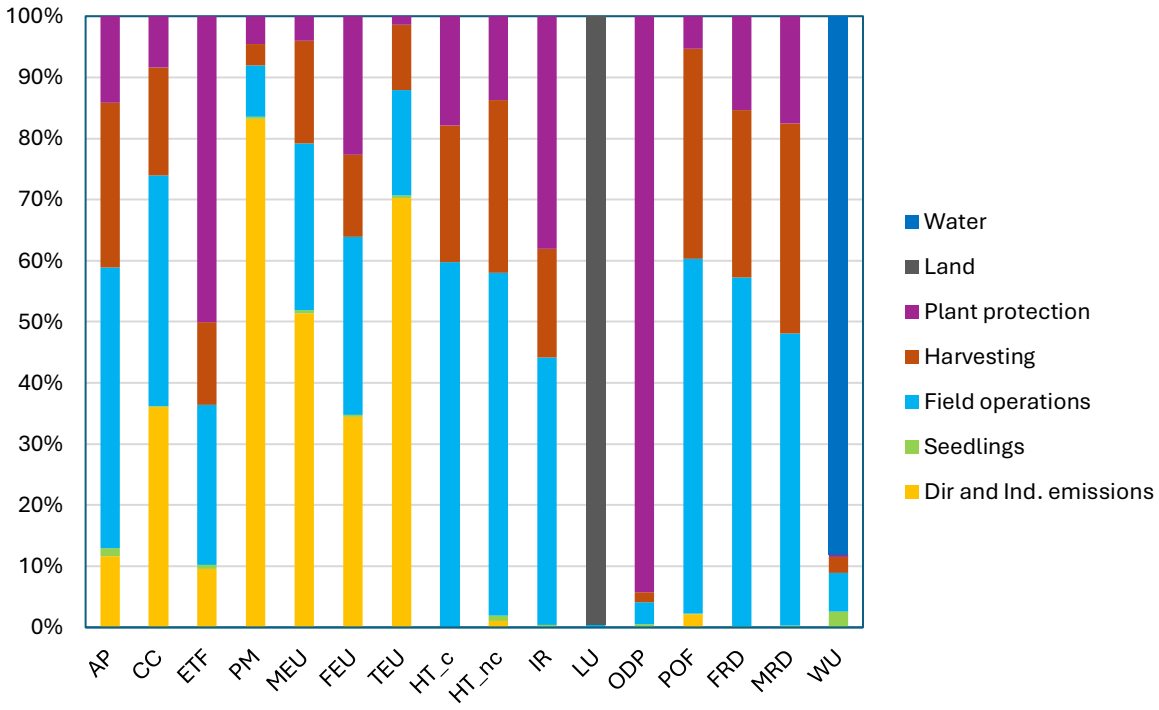



**Figure 4.8.** Relative contribution analysis (spinach – organic production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### *4.4.4 Tomato*

Table 4.5 presents the absolute midpoint impact assessment scores for 1 t of tomato production, with a corresponding contribution analysis in Figure 4.9. Tomatoes had the best environmental performance among the assessed crops, though direct comparisons are challenging due to differences in crop function and market sectors. However, the impact scores were relatively low compared to other horticultural products, primarily attributed to the limited and low input requirements coupled with a high yield of about 55 t/ha. Regarding relative contribution, emissions from agrochemicals and fertilizers were the primary drivers of most impact categories, except for land use, ozone depletion, and water use. For climate change, the total CC score was 55 kg $CO_2$ eq./t, with agrochemical and fertilizer emissions contributing equally at 28% each. This CC result is

consistent with findings from other studies, which reported around 50 kg $CO_2$ eq./t [206, 207]. However, higher scores of 100 and 500 kg $CO_2$ eq./t were reported for open-field tomato production in Greece for industrial and fresh markets, respectively [208].

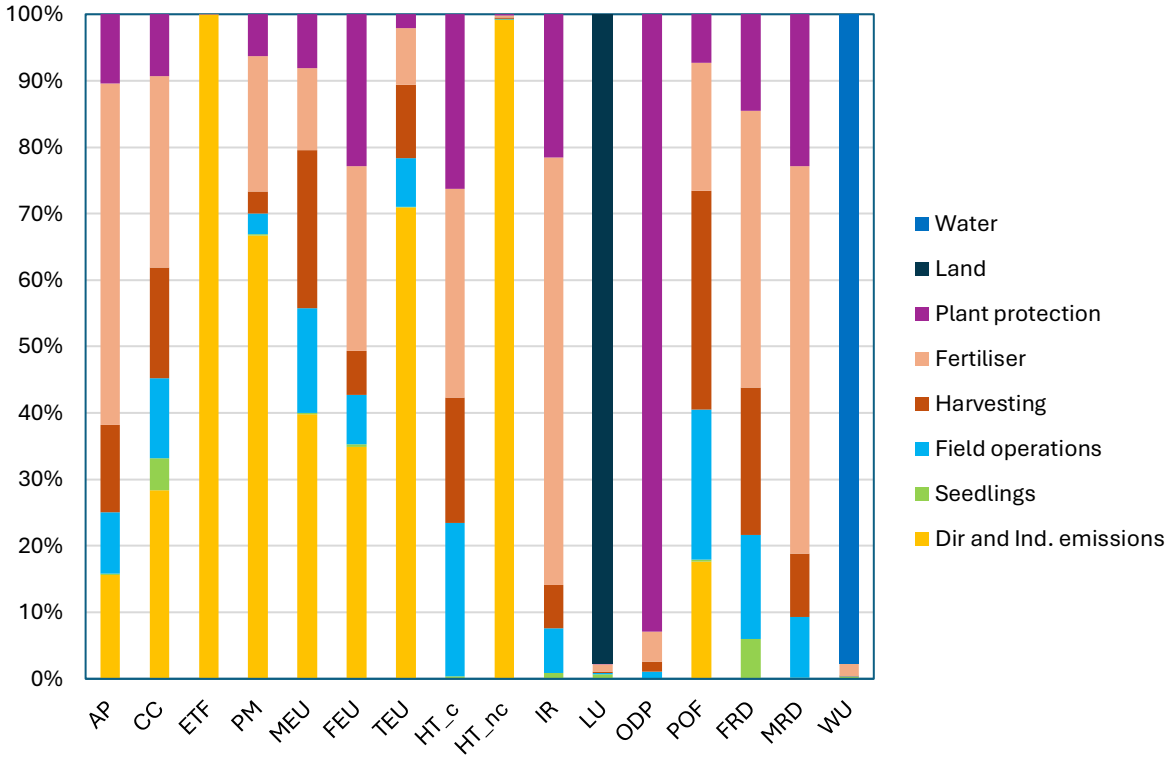



**Figure 4.9.** Relative contribution analysis (tomato – integrated production). (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

## 4.5 CONCLUSIONS

This study provides an overview of the environmental performance of the cultivation phase within the horticultural supply chain. To this end, a life cycle assessment of the production of four crops (onion, pea, spinach, and tomato) cultivated under integrated and organic production systems (except in the case of tomato, where only integrated was assessed) at the farm gate was conducted. The LCI was derived from primary data from an Italian agricultural consortium company primarily operating in central Italy.

The impact assessment revealed that integrated farming systems had a relatively higher environmental burden than organic systems across all evaluated impact categories. This outcome is attributed to the increased agrochemical inputs and associated emissions in integrated farming. However, the absence of specific processes for organic agricultural products introduces potential bias, favoring organic systems. Future research is needed to provide characterization factors for these products for balanced comparisons. Additionally, excluding organic farms with zero yields due to biotic or abiotic factors can skew the results, as it impacts environmental performance and has economic implications. It is also worth noting that, in organic farming, farmers often utilize higher-quality land with better soil and climate conditions due to restrictions on agrochemical use. In contrast, integrated farming systems may be carried out on lower-quality land, relying on agrochemicals to compensate for limitations. This difference in land allocation strategy also accounts for the perceived better environmental performance of the organic systems than the integrated systems. However, this complicates direct comparisons between the two systems. Again, the small sample size of organic farms (less than 10%, except for onion cultivation) compared to integrated farms increases uncertainty and potential bias in the analysis. A larger sample size of organic farms is needed for more robust conclusions regarding the relative sustainability of these production systems. Therefore, selecting the most environmentally sustainable farming system should consider factors like cultivar, soil, and climatic conditions influencing productivity.

Regarding environmental hotspots, the key contributors varied across crops and impact categories. The primary contributors, especially in integrated systems, included direct and indirect emissions from agrochemicals (notably $N_2O$ and NO), fertilizer production, and seedling production. Strategies such as reducing fertilizer use, improving nitrogen use efficiency, and incorporating nitrification inhibitors could significantly reduce horticultural impacts. Farmers should also be encouraged to exploit the nitrogen-fixing abilities of crops like peas to minimize nitrogen inputs. Optimizing efficiency and reducing impacts during the cultivation phase are crucial steps toward improving the environmental sustainability of frozen horticultural products. These efforts align with broader goals to promote sustainable diets.

# 5 PROCESSING HORTICULTURAL PRODUCE

## 5. 1 CHAPTER SUMMARY

This chapter covers the application of LCA to evaluate the environmental impacts of processing fresh vegetables into frozen products at C.O.VAL. M. – O.R.T.O. Verde S.c.a.p.a. Specifically, the study focuses on the processing of spinach and peas, as data limitations prevented the assessment of other horticultural products. The study covers the post-harvest and processing stages, adopting a farm-gate to factory-gate approach. The analysis aims to identify key environmental hotspots and opportunities for improvement within the processing stage.

An attributional LCA was conducted using the Environmental Footprint Method 3.1 to assess multiple midpoint impact categories. Primary data collected between 2019 and 2021 provided detailed information on transportation, energy consumption, packaging materials, and waste generation.

Electricity consumption emerged as the primary driver of environmental impact for both crops, primarily due to its extensive use in freezing, conveyor belts, and lighting. Natural gas and packaging materials also contributed significantly to certain impact categories. Furthermore, a significant challenge in the processing of spinach and peas is the generation of substantial amounts of residue, ranging from 15% to 30% of the initial input. Thus, spinach processing exhibited a higher overall environmental burden than pea processing, primarily due to greater processing losses and associated waste generation.

Several strategies, such as reducing waste generation and optimizing energy consumption through technological advancements and process improvements, can be implemented to enhance the environmental performance of frozen vegetable production. Moreover, sustainable packaging materials should be adopted, packaging waste should be reduced, and end-of-life management through reuse and recycling should be improved. Exploring innovative ways to utilize processing residues, such as bioconversion using BSFL, to recover valuable nutrients and reduce waste could also reduce impacts. The frozen vegetable industry can significantly reduce its environmental impacts and contribute to a more sustainable food system by addressing these key areas.

## 5.2 INTRODUCTION

Global food production and processing are rapidly expanding due to population growth and evolving lifestyles, leading to increased consumption of natural resources [159, 209]. Food processing plays a pivotal role in modern agriculture and food supply chains by enhancing food products' availability, safety, and convenience [210]. As a significant value-adding sector, food processing contributes substantially to economic growth [211]. It plays a crucial role in ensuring food security and meeting consumer demand [212]. However, it also significantly contributes to environmental issues, including greenhouse gas emissions, water consumption, and waste generation [212]. Therefore, there is a need for food manufacturing companies to transition to more environmentally sustainable practices.

Promoting sustainable food processing involves operating with greater efficiency, reducing energy consumption, minimizing waste, and extending the shelf life of food products [212, 213]. However, a critical step toward improving the sector's environmental performance is the comprehensive assessment of its impacts. LCA is a valuable tool for assessing the environmental performance of food products and processes [38, 39]. By analyzing the entire life cycle, from raw material extraction to end-of-life disposal, LCA can identify hotspots and opportunities for improvement. This information can inform decision-making, help in the development of more sustainable practices, and promote sustainable food processing [214].

Freezing is a common food processing technology that can preserve nutritional value, reduce food waste, and meet the growing demand for convenience [215, 216]. Freezing extends the shelf life of vegetables while retaining their freshness and nutrients, making them a suitable alternative to fresh produce [216]. Frozen vegetables also ensure the year-round availability of seasonal crops, bolstering food security and reducing dependence on imports. The convenience of frozen vegetable has made them integral to households, food service industries, and manufacturing, especially as modern lifestyles demand quick and nutritious meal options [212, 216].

Despite these advantages, frozen vegetable production poses environmental challenges, underscoring the need for sustainable practices throughout the value chain. Addressing

these challenges requires assessing the environmental impact of processing to identify hotspots and implement targeted improvement initiatives. Thus, this study aims to quantify the environmental impacts of frozen vegetable production by a supplier in central Italy, using LCA to highlight environmental hotspots and explore opportunities for enhancing sustainability.

## 5.3 MATERIALS AND METHODS

### *5.3.1 Goal and scope definition*

This LCA study aimed to evaluate the environmental impacts of processing horticultural products into frozen products by C.O.VAL. M. – O.R.T.O. Verde S.c.a.p.a. The primary objective was to identify critical environmental hotspots within the processing phase and establish a baseline for improving the environmental sustainability of the Italian frozen horticulture production chain.

The LCI modeling follows an attributional framework, focusing on how the production of frozen horticultural products influences environmental impacts and does not evaluate the consequences of frozen horticultural product consumption [46]. It is a product decision support study. Thus, the decision context for this study aligns with Situation A (micro-level decision support) as defined in the ILCD Handbook [40].The results are intended to inform decision-making at the processing level and within the horticultural supply chain without necessitating structural changes in the horticultural supply chain. The findings are aimed at a target audience comprising horticultural processors, consumers, farmers, agricultural experts, and policymakers to support strategies to reduce the environmental impact of the frozen vegetable supply chain.

The function of the system is to produce and sell frozen horticultural products from the processing of fresh horticultural products produced mainly in the central parts of Italy. Therefore, the selected FU for this study is the production of 1 t of frozen horticultural product at the factory gate, excluding packaging weight at the company gate in 2019 – 2022. The system boundary encompasses energy and material inputs and outputs associated with producing packaged frozen vegetables, as depicted in Figure 5.1. The study focuses on the post-harvest and processing phases, excluding cultivation, retail

storage, consumer transportation, product use, and packaging end-of-life. Due to data limitations, the analysis did not include infrastructure and the transportation of input materials (e.g., packaging). Additionally, cleaning agents were excluded as the processor indicated they were not used directly on processing lines. This study adopts a "farm gate-to-factory gate" approach.

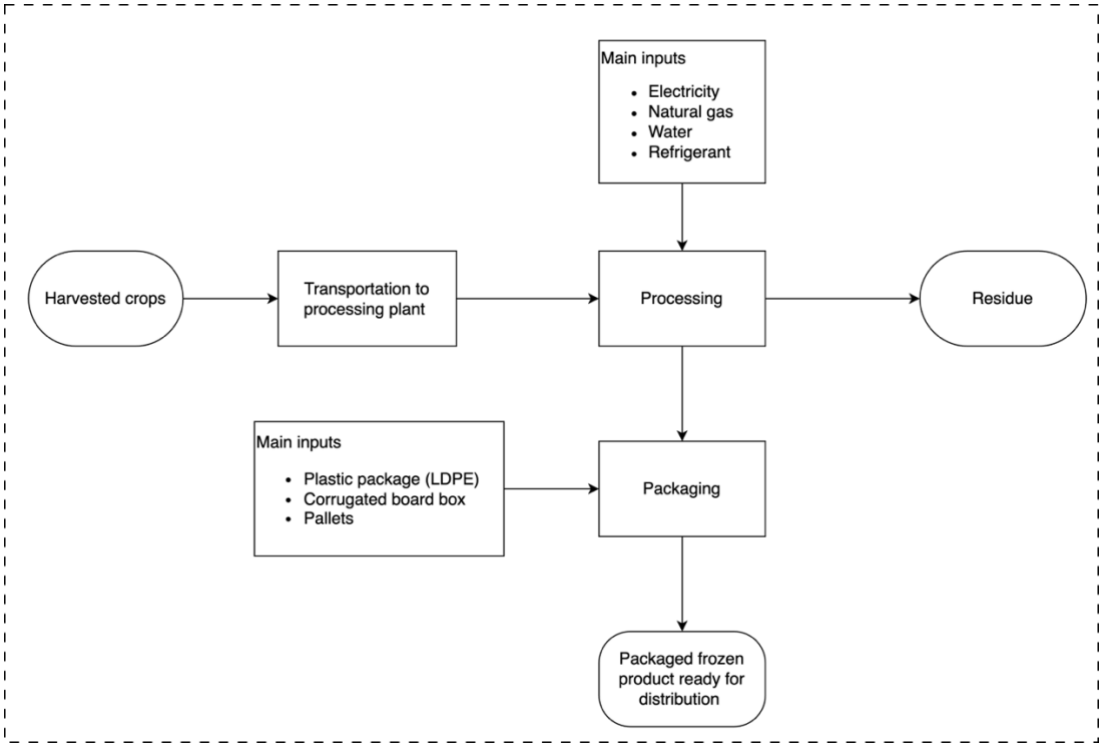



**Figure 5.1** System boundary for processing phase.

*5.3.2 System description and LCI modeling*

5.3.2.1 Spinach

Primary data was collected for the reference year 2019 to conduct the LCA. This data provided detailed information on the specific processes and energy consumption associated with the production system. Approximately 30% of the fresh spinach is lost during processing, requiring 1.42 kg of fresh spinach (reference flow) to produce 1 kg of frozen spinach.

Spinach is processed right after harvesting and transported under optimal conditions. The harvested spinach is transported to the processing facility by truck. The type of truck and transportation distance vary depending on the origin of the spinach. Spinach cultivated in the Marche region is typically transported using trucks with an 11-ton payload, while spinach from other areas is transported using trucks with a 23-ton payload. The trucks are diesel-powered and operate with a load factor of 50% to prevent damage due to spinach's low density and compressive strength. For the return journey, the trucks travel empty as they are exclusively dedicated to transporting fresh vegetables.

Upon arrival, the spinach undergoes several cleaning and sorting processes, including removing sand, stones, and defective produce. The cleaned spinach is then cooked in a multi-phase process involving heating and cooling. After cooking, the spinach is cut into cubes and frozen using ammonia as a refrigerant. Some products may undergo a glazing process to improve their appearance and prevent damage. The processing facility primarily uses electricity and natural gas to power its operations. Electricity is used to power conveyor belts, pneumatic systems, refrigeration equipment, water pumps, and lighting. Natural gas is used to heat the cooking process and provide heating for the facility.

As shown in Figure 5.2, fresh spinach undergoes minimal processing to produce the final packaged frozen product. Upon arrival, the spinach undergoes several cleaning and sorting processes, including removing sand, stones, and defective produce. The first steps involve three separators (sand trap, pneumatic separator, and optical separator) to remove impurities like sand, stones, stems, and defective spinach. The spinach is then washed in two stages using purified well water, and it is then cooked in a multi-phase process involving heating and cooling stages with four temperature phases: 75°C (pre-heating), 90°C (heating), 80°C (pre-cooling), and 23°C (cooling). After visual inspection, any rejected material is sent to the biogas plant. After cooking, the spinach is cut and pressed into cubes, frozen at -35°C using ammonia as the refrigerant in a closed system and optionally glazed to enhance the appearance and reduce damage. The processing facility primarily uses electricity and natural gas to power its operations. Electricity from the national grid is used for powering conveyor belts, pneumatic systems, refrigeration equipment, water pumps, and lighting. Natural gas is primarily used to heat the cooker

and offices. Due to limited data on energy use by specific processes, the study relied on cumulative electricity and natural gas data provided by the company.

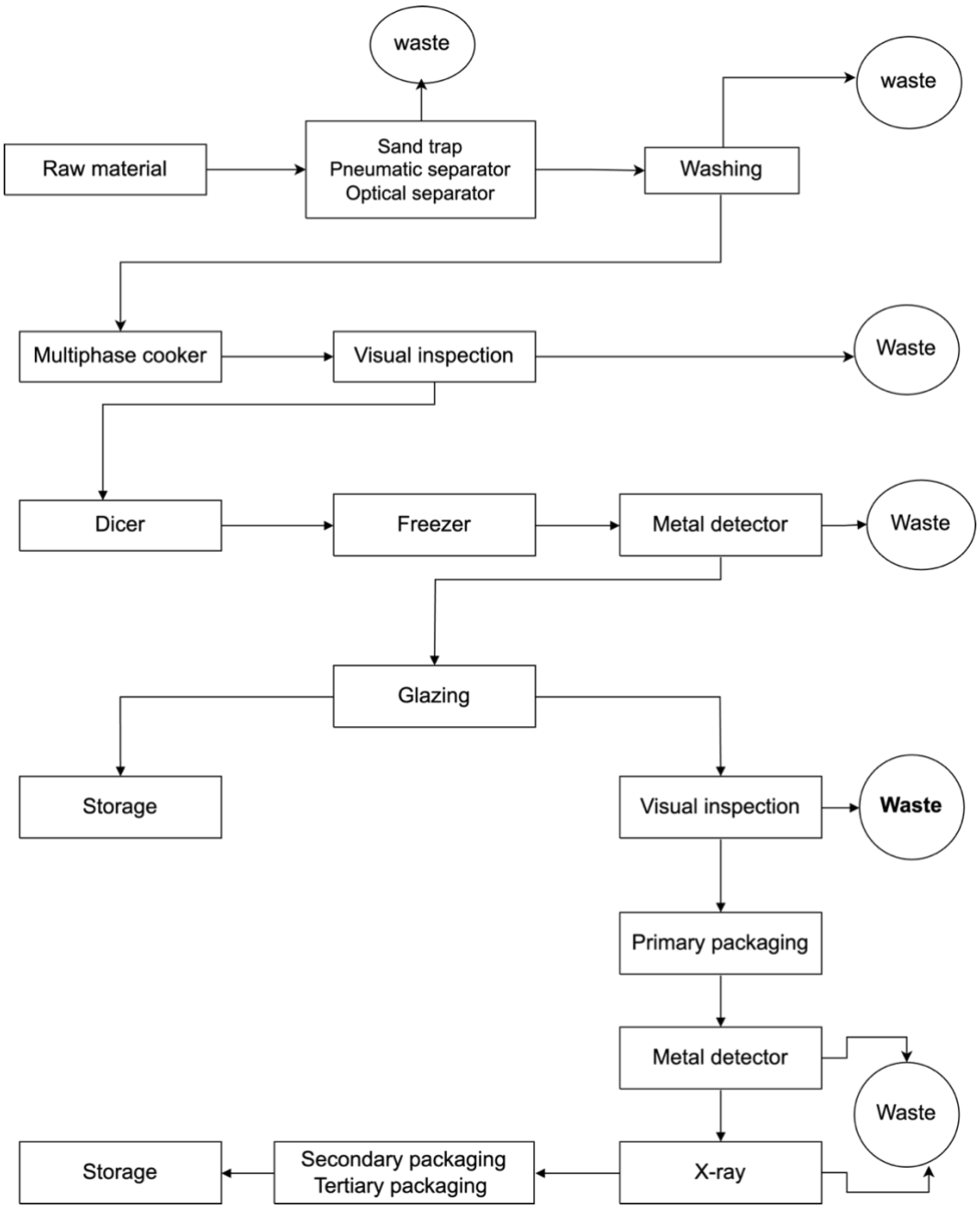



**Figure 5.2.** Flow diagram for spinach processing.

The packaging process includes multiple inspections to ensure product quality. After a visual inspection, spinach is packaged in plastic as the primary layer, followed by metal detection and X-ray quality checks. Products are protected with two packaging layers: plastic primary packaging and a cardboard secondary container for B-2-C products. For B-2-B products, bulk spinach is packaged in plastic bags weighing 1.09 kg for 650 kg of frozen spinach, with no use of cardboard boxes. Packaging bags for 10 kg and 20 kg weigh 9 g and 26 g, respectively. Pallets, typically wooden with dimensions of 80 x 120 cm, transport 54 to 81 carton boxes, depending on the product type. Pallets weigh approximately 23 kg. Frozen spinach is stored in freezing cells, with cubes in large cardboard boxes lined with plastic bags. Depending on customer demand, storage duration ranges from 2 days to 6 months. Refrigerant (ammonia) leakage was calculated based on standard rates: 1% during assembly, 8% annually during operation, and 5% at disposal (assuming 95% recovery). The refrigeration system's operational lifetime was estimated at 20 years.

The plant operates a wastewater treatment system to manage the effluent generated from the processing of various vegetables, including spinach, peas, tomatoes, and green beans. Due to the varying nature of the wastewater, depending on the specific processing steps involved, obtaining precise primary data on water quality proved challenging. Therefore, secondary data from the Ecoinvent database represented the wastewater treatment process, selecting a unit process that closely matched the plant's characteristics. This approach involved assuming that the wastewater was slightly organic-contaminated and did not require advanced treatment. Data for the spinach processing and packaging phases at the processing facility after cultivation are summarized in Table 5.1.

**Table 5.1**: The inventory data for producing frozen spinach at the processing facility. Data reported per kg of frozen product.

| Parameter | Unit | Amount |
|---|---|---|
| *Input* | | |
| Cultivated fresh spinach | kg | 1.42E+00 |
| Transported spinach | tkm | 3.12E-01 |
| Electricity | kWh | 5.19E-01 |
| Natural gas | $m^3$ | 6.14E-02 |
| Well water | kg | 1.77E+01 |
| Water purification | kg | 1.77E+01 |
| Refrigerant ($NH_3$) | kg | 2.10E-05 |
| *Packaging* | | |
| Plastic (low-density polyethylene) | kg | 1.05E-02 |
| Corrugated box | kg | 3.85E-02 |
| Printed paper labels | kg | 1.73E-04 |
| Pallets (wood) | kg | 1.70E-03 |
| *Outputs* | | |
| Frozen spinach | kg | 1.00E+00 |
| Spinach residue | kg | 4.18E-01 |
| *Emissions* | | |
| Refrigerant ($NH_3$) | kg | 3.03E-06 |
| Wastewater treatment | kg | 1.41E+01 |

5.3.2.2 Green peas

Regarding pea processing, primary data was collected for the reference year 2020 – 2021. This data provided detailed information on the specific processes and energy consumption associated with the production system. About 15% of the fresh pea is lost as residue during processing. Therefore, 1.18 kg of fresh peas (reference flow) is required to produce 1 kg of frozen peas.

The harvested peas are transported to the processing plant immediately after harvest to minimize quality loss. Third-party transporters use diesel-powered trucks with a load capacity of 7 tons to transport the peas. To prevent spoilage, the trucks are open-top and operate at 60% load capacity to avoid mechanical damage. The average transport distance ranges from 33 to 357 km. As the trucks are dedicated to food transportation, they typically return empty.

Figure 5.3 outlines the transformation of fresh peas into packaged frozen products through minimal processing. The process starts with four separation steps (pneumatic separator, water stone remover, vibrating screen, and belt stone remover) to remove impurities such as sand, stones, insects, and other debris. The peas are then washed using two water-based processes, fluctuation and decantation, with water sourced from artesian wells. Blanching follows, where peas are steamed in a rotating drum cooker, heated to 94°C, and then cooled to 23°C. After a visual quality inspection, defective parts are removed and discarded as residue. Wastewater from the process is treated in a dedicated purification system. The final product is frozen and stored at -35°C using ammonia as the refrigerant in a closed system. Electricity, sourced from the national grid and a co-generator, powers conveyor belts, the pneumatic system, freezers, compressors, water pumps, electric forklifts, and lighting. Natural gas is used primarily for heating the steam cooker and office heating. Primary data from the reference year was used to quantify natural gas and electricity consumption. Details of the pea processing and packaging phases after cultivation are summarized in Table 5.2.

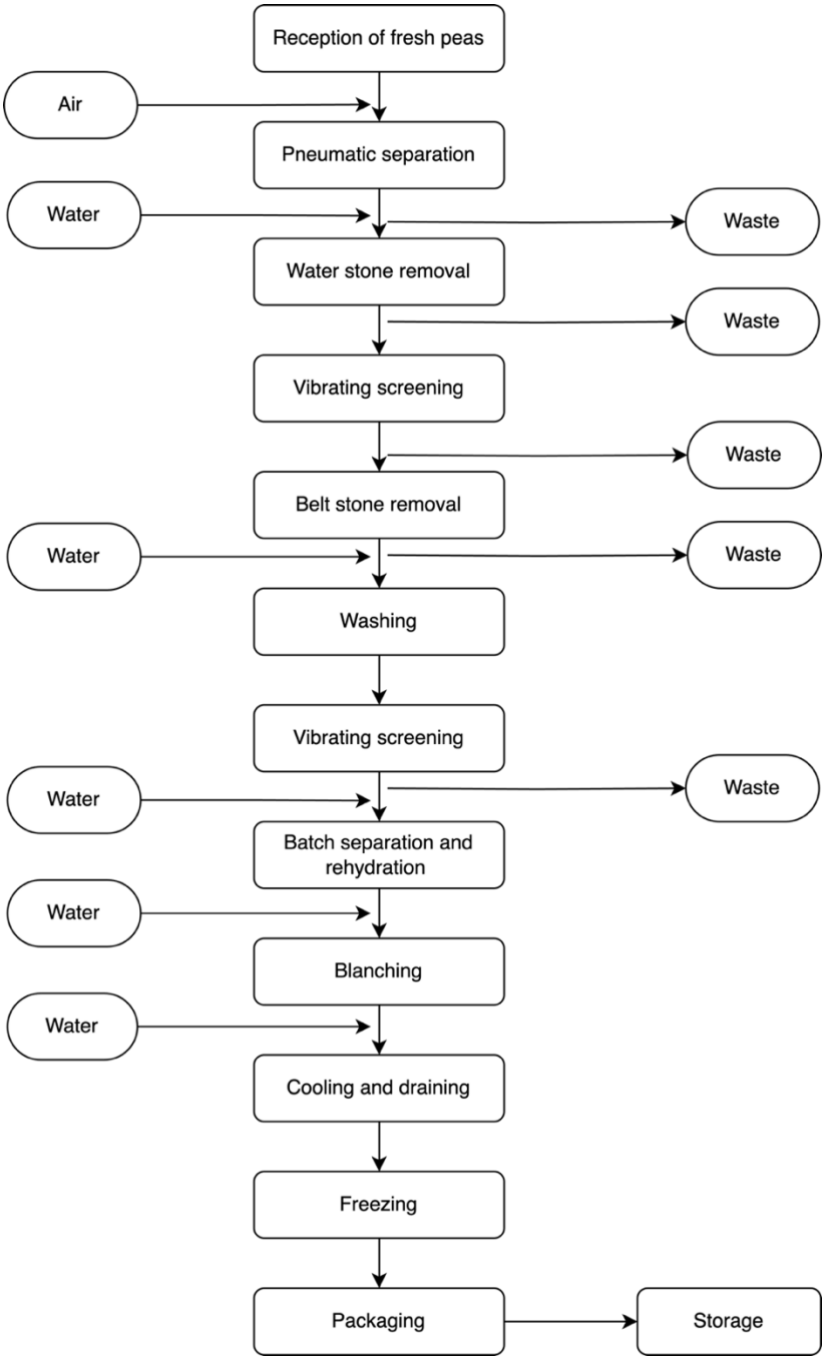



**Figure 5.3.** Flow diagram for pea processing.

The products are protected and transported using two main packaging layers for wholesale distribution. The primary layer is a plastic bag, which prevents contamination, while the secondary layer is a carton box for structural support and handling. Standard wooden pallets (100 x 120 cm) are used for transportation, with an average weight of 30 kg and an estimated lifespan of 5 years. Pallet usage was modeled based on company data.

However, the transport distances for packaging materials were excluded due to lack of information. Frozen peas are stored in freezing cells in large cardboard boxes lined with plastic bags, with storage durations ranging from 2 days to 6 months based on customer demand. The company primarily supplies bulk frozen peas to large companies for final packaging. Refrigerant (ammonia) leakage was calculated using recommended rates: assembly (1%), annual leakage from operation (8%), and leakage from dumped refrigeration equipment (5%, assuming 95% recovery). The operational lifetime of the refrigeration system was estimated at 20 years.

**Table 5.2.** The inventory data for producing frozen pea at the processing facility. Data reported per kg of frozen product.

| **Parameter** | **Unit** | **Amount/FU** |
|---|---|---|
| *Input* | | |
| Fresh peas (from the farm) | kg | 1.18 |
| Transport (farm to factory) | kg-km | 1.56E-02 |
| Electricity (grid) | kWh | 0.12 |
| Electricity (co-generator) | kWh | 0.13 |
| Natural gas | MJ | 0.68 |
| Well water | kg | 17.7 |
| Water purification | kg | 17.7 |
| Refrigerant ($NH_3$) | kg | 2.43E-05 |
| Plastic packaging (LDPE) | kg | 1.05E-03 |
| Plastic bin (polypropylene) | kg | 8.85E-04 |
| Corrugated board box | kg | 5.12E-03 |
| Pallet | p | 8.41E-04 |
| Iron mesh cage | kg | 5.63E-03 |
| *Output* | | |
| Frozen peas | kg | 1 |
| Pea residue | kg | 0.18 |
| *Emissions* | | |
| Refrigerant ($NH_3$) | kg | 3.28E-06 |
| Wastewater | kg | 1.41E+01 |

*5.3.3 Interpretation*

The absolute midpoint characterization results for each tonne of frozen product are presented. Results are reported across multiple impact categories, focusing on climate change. A detailed analysis of the contributing phases, key processes, and substances (hotspot analysis) is provided to interpret the findings.

## 5.4 RESULTS AND DISCUSSION

This LCA study evaluated the environmental impacts of processing spinach and peas into frozen products from the farm gate to the factory gate. The results in Table 5.3 indicate that pea processing had a lower environmental impact than spinach processing across all impact categories. For example, the CC impact for the spinach process (415 kg $CO_2$ eq./t product) was nearly four times that of pea processing (110 kg $CO_2$ eq./t product).

This difference in environmental performance can be attributed to the higher processing losses associated with spinach. Approximately 30% of fresh spinach is lost during processing, equating to 1817 t of waste, compared to 15% loss (1342 t) for peas. These losses, driven by strict quality standards and food safety requirements, result in the disposal of produce with minor defects. Therefore, strategies aimed at improving processing efficiency and minimizing waste generation are crucial to reduce the environmental burden. Additionally, exploring innovative ways to valorize processing residues could further enhance the environmental performance of frozen vegetable production.

**Table 5.3.** Farm gate-to-factory gate LCIA results for frozen horticultural products (reported per t of frozen product).

| Impact | Unit | Frozen spinach | Frozen pea |
|---|---|---|---|
| AP | mol $H^+$ eq. | 1.67 | 0.42 |
| CC | kg $CO_2$ eq. | 415.1 | 110.21 |
| ETF | CTUe | 1831.55 | 392.69 |
| PM | disease inc. | 1.67E-05 | 4.31E-06 |
| MEU | kg N eq. | 0.42 | 0.10 |
| FEU | kg P eq. | 0.09 | 0.03 |
| TEU | mol N eq. | 3.98 | 0.97 |
| HT_c | CTUh | 2.89E-07 | 2.03E-07 |
| HT_nc | CTUh | 4.86E-06 | 1.52E-06 |
| IR | kBq U-235 eq. | 35.22 | 9.69 |
| LU | Pt | 1.11E+04 | 4.28E+03 |
| ODP | kg CFC11 eq. | 1.22E-05 | 3.59E-06 |
| POF | kg NMVOC eq. | 1.76 | 0.46 |

| | | | |
|---|---|---|---|
| FRD | MJ | 8.32E+03 | 2.18E+03 |
| MRD | kg Sb eq. | 2.89E-03 | 7.43E-04 |
| WU | $m^3$ depriv. | 5.60E+02 | 4.30E+02 |

(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### *5.4.1 Hotspots analysis*

#### 5.4.1.1 Spinach

Figure 5.4 presents a hotspot analysis of spinach processing, highlighting the contributions of various system components (e.g., electricity, natural gas, packaging) to the overall environmental impact across different categories. The contributions of each element varied significantly depending on the impact category. For example, pallets contributed 3% of CC but accounted for 70% of LU. Similarly, electricity accounted for 48% of the total CC impact and 9% of LU.

Electricity consumption emerged as the most significant contributor to many environmental impacts. Electricity is used extensively throughout the plant, powering conveyor belts, pneumatic systems, freezers, compressors, water pumps, electric forklifts, and lighting. The primary electricity data included the cumulative energy use of the plant, encompassing activities not directly related to spinach processing. Segregating energy data specific to spinach processing could potentially reduce the assessed impacts. Moreover, the technology used in the plant can be considered average, indicating lower efficiency in equipment such as conveyor belt engines. Implementing regular maintenance and upgrading equipment could improve efficiency and reduce environmental burdens.

Other system components also had notable contributions. Packaging accounted for 10 – 34% of impacts, particularly in MEU and FEU. Transportation significantly contributed to CC (18%), EFT (33%), and PM (32%). Natural gas, primarily used for heating cookers and offices, contributed to ODP (38%), FRD (30%), and POF (21%). Refrigerant use

contributed less than 1% across all impact categories. Wastewater treatment provided WU with environmental credit by recovering water from processing.

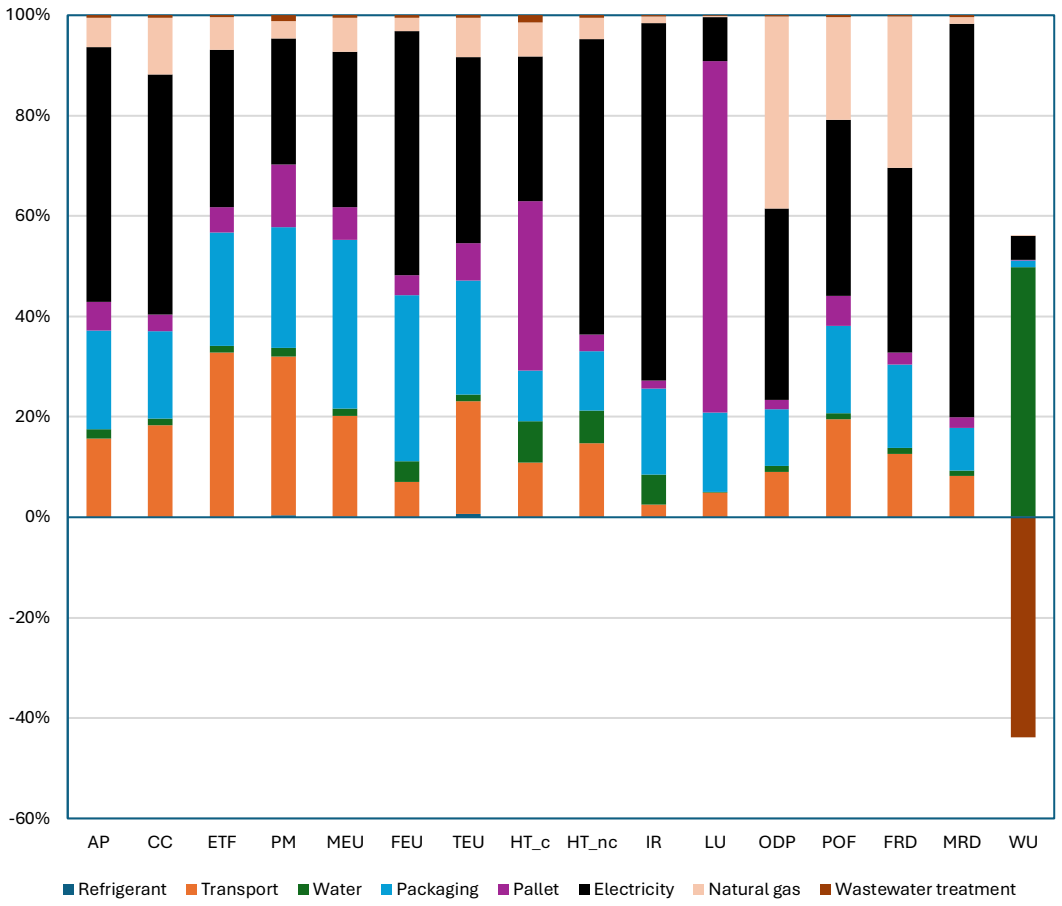



**Figure 5.4.** Hotspot analysis of the elements of the system contributing to the environmental impacts of spinach processing. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

When comparing these results to previous studies, Frankowska et al. [200] assessed the environmental impacts of vegetable consumption in the UK on a life cycle basis, including comparisons between locally produced and imported vegetable products. This is the only study to evaluate frozen spinach's environmental impacts. The current study's findings align closely with those reported by Frankowska et al. [200]. For instance, the CC impact

for processing fresh spinach into frozen spinach in the UK was 330 kg $CO_2$ eq./t product, while imported frozen spinach at the factory gate had a CC score of 377 kg $CO_2$ eq./t. In comparison, this study reported a slightly higher CC score of 415 kg $CO_2$ eq./t for frozen spinach.

These findings underscore the importance of addressing inefficiencies, especially in electricity consumption and packaging, while exploring strategies for waste valorization to improve the environmental performance of spinach processing.

5.4.1.2 Green peas

The hotspot analysis for pea processing, showing the contributions of various elements to the assessed environmental impacts, is presented in Figure 5.5. The results were comparable to those observed for spinach processing, with the most impactful elements varying across the impact categories. Electricity, natural gas, and the iron cage were identified as the primary drivers of environmental impacts.

Electricity was the most significant contributor to CC (54%), AP (50%), IR (61%), and MRD (72%). Natural gas significantly contributed to ODP (39%), FRD (34%), and POF (23%). The iron mesh cage accounted for 48% of HT_c, while the wooden pallet was the dominant contributor to LU (87%). Unlike spinach processing, transportation had a negligible impact, contributing less than 2% across all categories assessed. Like spinach processing, wastewater treatment provided environmental credit for WU due to water recovery during processing.

Comparing these findings with previous studies, the results align with earlier research on pea processing. In this study, the CC impact for pea processing was 110 kg $CO_2$ eq./t frozen pea, which is lower than the values reported by Del Borghi et al. [217], who found CC impacts of 250 kg $CO_2$ eq./t for steam-cooked peas and 150 kg $CO_2$ eq./t for peas without packaging. However, Frankowska et al. [200] reported substantially higher CC impacts of 960 kg $CO_2$ eq./t for frozen shelled peas produced in the UK and 810 kg $CO_2$ eq./t for frozen shelled peas imported into the UK. These comparisons highlight the variability in environmental impacts due to differences in processing methods, geographic

contexts, and system boundaries, underscoring the importance of identifying specific inefficiencies to improve the sustainability of pea processing.

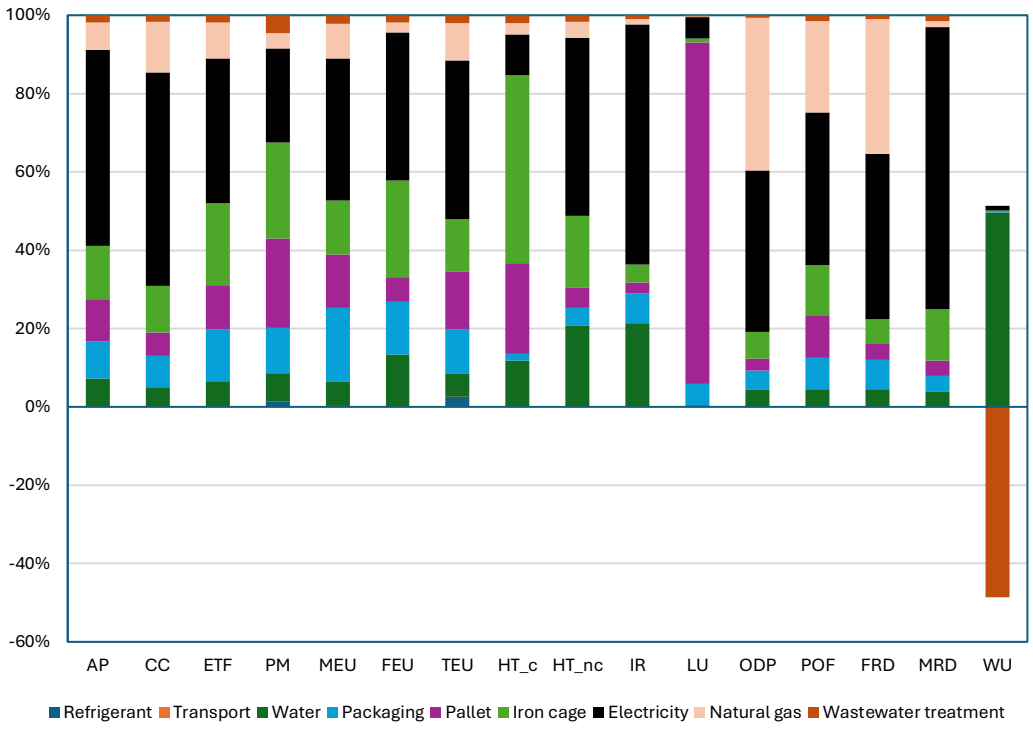



**Figure 5.5** Hotspot analysis of the elements of the system contributing to the environmental impacts of pea processing. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

## 5.5 CONCLUSIONS

This LCA study assessed the environmental impacts of frozen pea and spinach production from farm gate to factory gate using the LCA. The results highlight the importance of identifying and addressing critical environmental impacts to improve the sustainability of horticultural frozen output. Electricity consumption emerged as the primary driver of environmental impact for spinach and pea processing, contributing significantly to categories such as CC, AP, FRD, and MRD. Natural gas consumption and

packaging materials also played significant roles, particularly in ODP, FRD, and specific toxicity categories.

Spinach processing exhibited a higher overall environmental burden than pea processing, partly due to more significant losses during processing and the associated waste generation. Pea processing demonstrated relatively lower impacts across most categories, with transportation contributing minimally to the overall environmental impacts.

Comparisons with previous studies revealed that the environmental performance of spinach and pea processing in this study aligns closely with global benchmarks. However, variations exist due to differences in system boundaries, processing technologies, and regional contexts. These findings emphasize the potential for significant improvements in environmental performance through targeted interventions, such as upgrading equipment for better energy efficiency, reducing waste during processing, and exploring innovative valorization pathways for by-products like residues. By addressing these challenges, the sustainability of frozen vegetable production can be enhanced, supporting the growing demand for environmentally sustainable food production systems.

The next chapter will explore the potential of valorizing processing residues through bioconversion using BSFL as a strategy to enhance the environmental performance of the frozen vegetable value chain.

# 6 RESIDUE VALORIZATION WITH BSFL

## 6.1 CHAPTER SUMMARY

This chapter applies LCA to evaluate the environmental performance of BSFL production using diets composed of agro-industrial residues. These residues come from the processing of frozen horticultural products by C.O.VAL. M. – O.R.T.O. Verde S.c.a.p.a. The analysis explores the potential of BSFL rearing to valorize these residues while addressing key sustainability concerns.

The specific objectives of this chapter are to:

- Assess the environmental performance of BSFL production on various diets.
- Identify key environmental hotspots in BSFL production.
- Compare the environmental impacts of BSFL production with conventional protein sources for animal feed.
- Compare the environmental impacts of BSFL production with conventional lipid sources for bioenergy production.
- Discuss the opportunities and challenges of introducing BSFL production as a nutrient recycling method.

An attributional LCA was conducted using primary data collected in 2024 from the rearing system to achieve these objectives. The Environmental Footprint Method 3.1 was employed to assess multiple midpoint impact categories.

The results reveal that agro-industrial residues and liquid digestate are effective substrates for BSFL production. However, the environmental impacts of BSFL production were substantially higher than those of conventional protein and plant lipid sources for animal feed and bioenergy production, respectively. The electricity used for climate control in the closed-rearing system was the primary contributor to these impacts.

Strategies to mitigate these impacts include improving energy efficiency, adopting cleaner energy sources, and valorizing waste heat from biogas plants. The location of the rearing facility plays a critical role in optimizing environmental performance, particularly in terms

of access to renewable energy and waste heat. Additionally, liquid digestate, a substrate often challenging to dispose of due to regulatory constraints, proved to be a viable feedstock for BSFL. Integrating BSFL production with anaerobic digestion can address the challenges associated with residue and liquid digestate disposal while benefiting from both systems. However, microbial safety concerns related to liquid digestate must be resolved.

By adopting these strategies, the horticultural industry can enhance the environmental sustainability of frozen vegetable production while contributing to more sustainable food systems. This chapter highlights the potential for BSFL production to support circular economy practices, particularly in nutrient recycling and residue valorization.

## 6.2 INTRODUCTION

The global population is rapidly growing, projected to reach over 9.6 billion by 2050. To meet the increasing demand for food and feed, global production must increase by over 70% [218]. Rising living standards, particularly among the middle class, have increased food consumption, especially animal protein. While agricultural expansion and intensification are necessary to meet this demand, they have significant adverse environmental impacts, including resource depletion and emissions of pollutants [219]. Furthermore, approximately one-third of all food produced globally is wasted, posing significant economic, environmental, and social challenges [2, 220]. Therefore, to simultaneously address the challenges of food production and waste management, recovering and upcycling nutrients from organic residues using the BSFL has emerged as a promising strategy. This solution can enhance the sustainability of agri-food production.

Insect farming, such as cricket and BSFL, offers a sustainable solution to global protein demand. BSFL efficiently converts organic waste into high-quality protein and other valuable products, reducing environmental impact and promoting circular economy principles[9, 218, 221]. BSFL production requires minimal water and land resources. Moreover, although they can digest various organic materials, they have a low risk of zoonotic disease transmission [222, 223]. With the recent lifting of the ban on Processed Animal Protein (PAP) for non-ruminant animals in 2021 by the European Commission, BSFL, recognized as a valuable PAP under Regulation EU 2021/1372, now has a broader

market potential [224]. It can be used in fish, pig, and poultry feed. Beyond protein production, the BSFL value chain offers additional economic opportunities, including bioenergy (e.g., biodiesel from accumulated lipids), biopolymers (chitin and chitosan), organic fertilizers, bioplastics, and bioactive compounds [13–15]. These products could replace conventional resources, contributing to a more sustainable and circular approach to biomass utilization.

Although the potential benefits of BSFL production are apparent, industrial-scale rearing, processing, and valorization come with some environmental and socioeconomic challenges [225]. Large-scale BSFL production, particularly during value-addition processes like protein and lipid extraction, can be resource and energy-intensive, potentially offsetting some of its sustainability advantages [36, 226]. Factors such as feed material sourcing, energy consumption, infrastructure, and post-harvest processing are critical in determining the overall environmental performance of BSFL production systems.

Therefore, a comprehensive evaluation of its life cycle is essential to support the sustainable commercialization of BSFL and maximize its environmental and economic benefits. This study aims to assess the environmental performance of BSFL production, focusing on diets based on agro-industrial residues and liquid digestate. By conducting an LCA, this research seeks to quantify environmental burdens, identify critical hotspots, and provide insights into strategies for improving the sustainability of BSFL-based bioconversion systems.

## 6.3 MATERIALS AND METHODS

To evaluate the potential environmental impacts of the studied BSFL production system, the guidelines provided in the ISO 14040/14044 standards [38, 39] and the International Reference Life Cycle Data System (ILCD) Handbook [40] were followed to conduct the LCA study. The LCA software SimaPro version 9.5. was used. Background LCI data was based on the attributional Ecoinvent database version 3.9 (allocation, cut-off by classification processes) [168].

### *6.3.1 Goal and scope definition*

The goal of the study is to evaluate the environmental impacts of BSFL biotreatment of agri-food industrial residue as a means to improve the overall sustainability of frozen horticultural products. Specifically, the study's objectives are to compare the environmental performance of BSFL on different agri-food industrial residues, identify key parameters of the biotreatment process, and assess their overall influence on environmental impacts through a sensitivity analysis.

Given the multifunctionality of the BSFL bioconversion system, several functional units were chosen;

- the production of 1 t of fresh BSFL biomass in a mass-scale prototype in 2024
- the provision of 1 kg of protein derived from BSFL in feed
- the provision of 1 MJ of energy from BSFL lipids

This attributional LCA study with substitution focuses on the BSFL biotreatment of agro-industrial residues (Figure 6.1). This study aligns with "Situation A: Micro-level decision support" as defined in the ILCD Handbook [40]. The objective was to assess the environmental impacts of utilizing BSFL to valorize agri-industrial residues while simultaneously producing fresh BSFL biomass. This micro-level focus implies that the study aims to inform decision-making at the processing level within the horticultural supply chain, without necessitating significant structural changes at the broader supply chain level. The findings of this study are intended to support decision-making by several stakeholders, including horticultural processors, waste management companies, farmers, agricultural experts, and policymakers. The insights from this research can be utilized to develop strategies for reducing the environmental impact of the horticultural supply chain by promoting the sustainable management of agro-industrial residues through BSFL bioconversion.

Given that the primary products of BSFL biotreatment are larvae (a protein and lipid source) and frass (a fertilizer), the system was expanded to account for potential substitutions with conventional products serving similar functions. This allowed for calculating the environmental credits associated with the avoided production of these alternative products. Primary data for this study were obtained from a mass-scale rearing

system located in Osimo, Marche region (Italy), near a biogas plant (C.O.Val.M. Biogas S.r.l.). The proximity of the BSFL prototype rearing system to the biogas plant facilitated access to fresh residues and the use of energy from the biogas plant. The reference year for the BSFL biotreatment data is 2024.

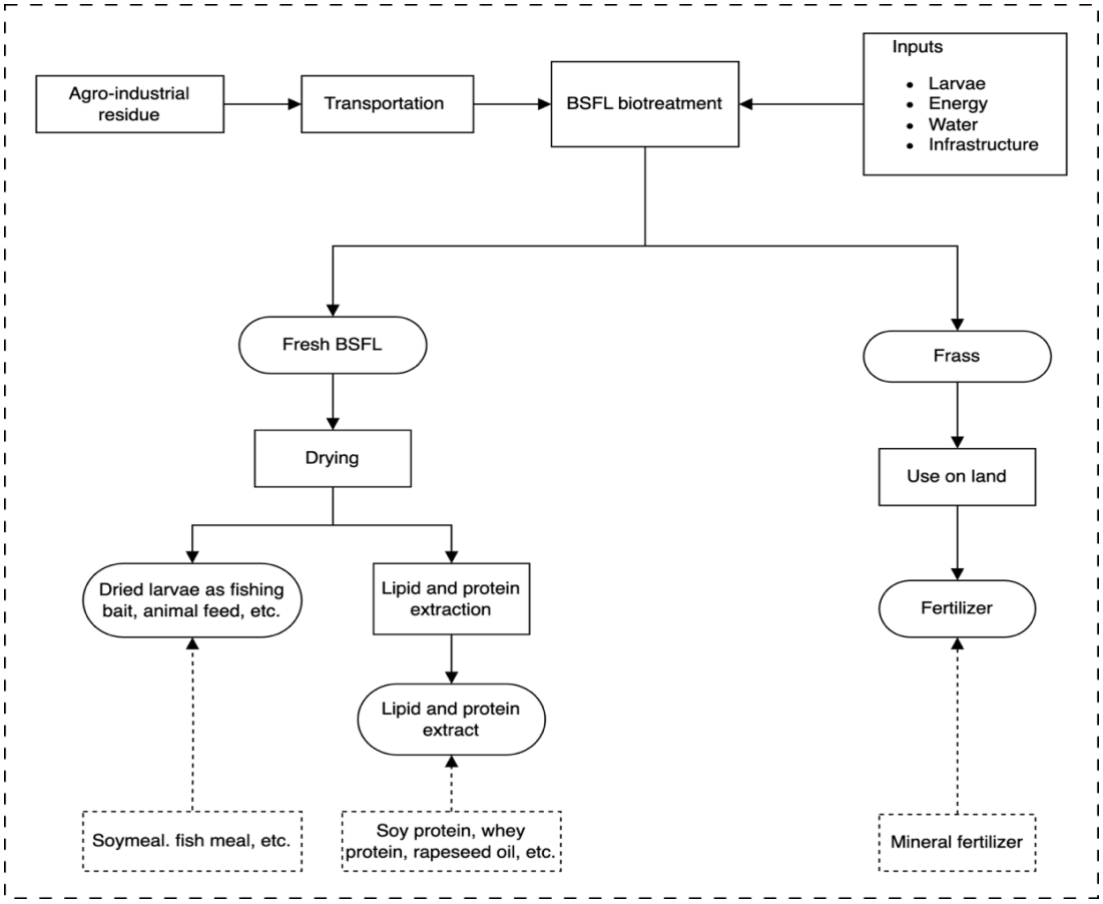



**Figure 6.1.** System boundary of the life cycle stages included in the study.

### *6.3.2 Description of the BSFL mass-scale rearing system*

The insect biotreatment was carried out in a newly constructed and dedicated pilot plant. The plant was a modified shipping reefer container with standard dimensions of approximately 12 m long, 2.4 m wide, and 2.5 m high, and a total useful volume of roughly 89 $m^3$ that can treat about 1.58 t of residues per cycle. Figure 6.2 shows the plant's basic rendering and layout, while Figure 6.3 (A – D) shows other images of the plant. The plant is divided into three sections: the bioconversion area, the area for feed and larvae processing, and the mating and reproduction area.

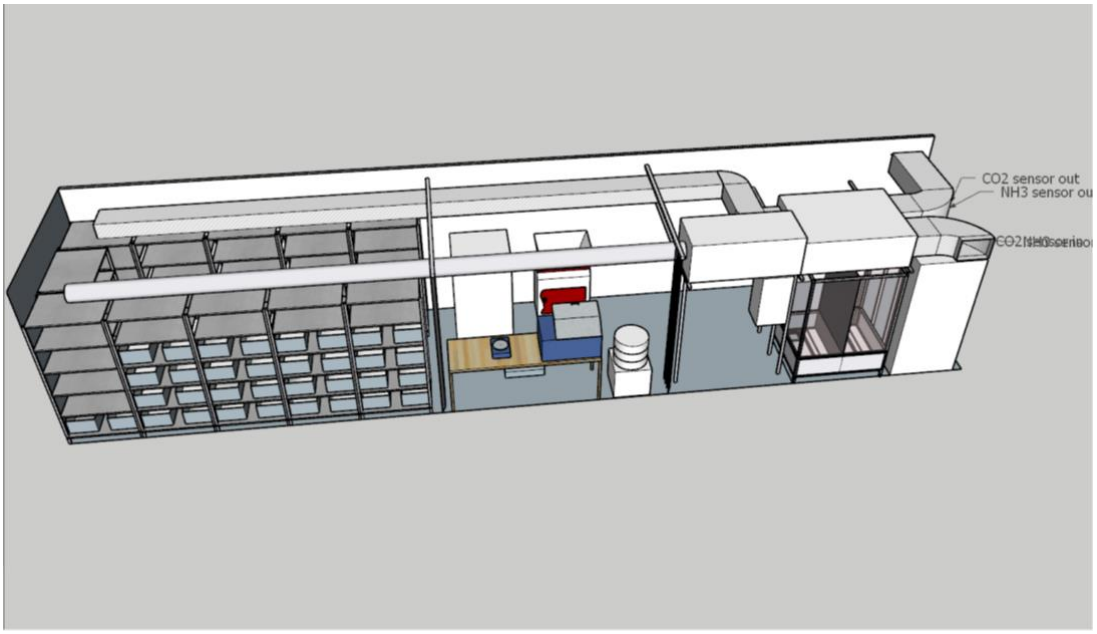



**Figure 6.2**. Basic rendering of the prototype. From left to right, separated by partitions, the breeding area, the processing area and the mating and reproduction area, respectively. The entrance is located on the far-right edge.

Inside the rearing facility, the residue is pre-treated by milling to reduce the particle size. The substrate is then weighed into a tray of dimensions 60 cm x 40 cm x 25 cm. To maintain optimal growing conditions for the BSFL, a climate control system has been implemented. A 10 kW heat pump, powered by renewable energy from a biogas plant, provides heating. Two 0.5 kW fans facilitate rapid heat distribution within the rearing chamber. Humidity levels are regulated using a humidifier, while a recuperator ensures adequate air exchange. The air exchange rate within the rearing chamber is maintained at 8 volumes/hour or 800 $m^3$/h, providing a well-ventilated environment for optimal larval development. The air delivery system incorporates a micro-perforated channel (Ø250 mm) with a length of about 8 m which ensures gentle and even air distribution. The air velocity within the rearing chamber remains low (<= 0.2 m/s at a distance of >= 1.2 m from the duct), allowing for flexible positioning of the adult fly aviary within the container. Both the larval and adult phases require temperatures ranging from 27 °C to over 30 °C with an optimum of 30 °C for adults. The optimal relative humidity is also around 70 – 80%, and this parameter is more limited for adults than for larvae. Harvesting is done by sieving. Other equipment installed in the plant for processing include, a mill (Tiger Shark

200S), a sieve (Guiliani tech. IG/1/S 500), a thermostatic bath (PID system M408-BM, MPM Instruments s.r.l.), and an oven (PID system M400VF, MPM Instruments s.r.l).
It is worth noting that the initial plan during the design of the prototype was to utilize residual heat from the biogas plant for climate control. However, technical and regulatory challenges hindered the implementation of this approach. As an alternative, an electric heat pump powered by electricity generated from the biogas plant was installed. This adjustment was considered a less favorable scenario economically and environmentally, especially when the electricity source is less sustainable or carbon-intensive.



(A) (B)

(C) (D)

**Figure 6.3**. The BSFL biotreatment facility. A – The external view with the heat exchanger, B – the control system, C – the bioconversion area and trays, D – the mating and hatchery unit.

*6.4.1 Data sources and inventory*

Experimental data was obtained from six trials conducted between May and August 2024. The residues evaluated in the study were categorized into two groups: fresh residues and dry residues. Fresh residues are characterized by high moisture content and short shelf life. This group included peas, onions, and tomatoes. Additionally, liquid digestate was tested in this category. In contrast, dry residues have very low moisture content and are relatively stable. This group included chickpeas from optical sorting, chickpeas from mechanical sorting, and wheat storage residues. The substrate formulations used for the BSFL diets tested are detailed in Table 6.1.

**Table 6.1.** Substrate formulation for *Hermetia illucens* production.

| Diet | Composition | Ratio |
|---|---|---|
| Diet 1 | Tomato + wheat | 2.53: 1 |
| Diet 2 | Onion + wheat | 3.5:1 |
| Diet 3 | Pea + chickpea | 3.5:1 |
| Diet 4 | Tomato + chickpea | 2.32:1 |
| Diet 5 | Liquid digestate + wheat | 2:1 |
| Diet 6 | Pea + wheat | 3.5:1 |

The residues were milled and mixed in the trays. Five-day-old BSF larvae obtained from – Smart Bug S.r.l (Ponzano Veneto, Italy) were added to the substrate. Each tray contained 12 kg of residue and 12,500 larvae, with a larvae density of 5 larvae/cm$^2$. The feeding was divided into two batches, 6 kg on day 1 and another 6 kg on day 5. The bioconversion takes 10 days for larvae to mature (5th instar stage). Afterward, harvesting was done with the sieve to separate the larvae and the frass.



(A)



(B)



(C)



(D)



(E)



(F)



(G)



(H)



(I) (J)

**Figure 6.4.** BSFL bioconversion activities at the rearing facility. A – diet preparation, B – residue milling, C – larvae, D and E – addition of larvae to diet, F – end of bioconversion, G – larvae separation from frass, H – sieving, I – frass, J – harvested larvae.

*6.4.2 Ambient condition monitoring*

Between September 2023 and April 2024, the breeding system was equipped with sensors to measure breeding parameters and emissions. This comprehensive monitoring system provided insights into emissions related to breeding conditions to facilitate improvements in environmental monitoring and the optimization of breeding processes. For the trials conducted in the prototype, low-cost sensors were chosen to enable continuous, real-time monitoring of gas fluctuations and gather useful data on system performance. Although high-cost sensors could offer greater accuracy, their high cost limited their feasibility for a prototype system. Using low-cost sensors provides a practical, cost-effective solution, balancing functionality and affordability. Considering these were study trials, the approach aligned with the goal of developing scalable and accessible solutions for broader applications. The goal was to address a significant gap in the literature, where previous studies often relied on laboratory-scale chamber systems to measure GHGs. Although useful for controlled experiments, these methods may not accurately represent real-world conditions and do not provide real-time data. Measurements at industrial or sub-industrial scales are also largely absent. As discussed in chapter 2, LCA studies on BSFL mostly rely on secondary data from less relevant systems or unrelated insect species due to a lack of specific data on this insect.

#### 6.4.2.1 $CO_2$ monitoring

Two twin sensors (S+S REGELTECHNIK AERASGARD KCO2-SD) were installed. These non-dispersive infrared (NDIR) optical sensors, equipped with automatic calibration, were strategically placed in the air inlet channel (before the heat pump filters) and the outlet channel (post-heat recovery). This configuration allowed for monitoring environmental $CO_2$ concentrations and the additional contribution of emissions from the breeding system. Given the proximity of the prototype to a biogas plant, the $CO_2$ sensors also evaluated potential interference from emissions such as those from the co-generator or the flare combustion torch. The sensors measured a range of 0–2000 ppm, with a precision of 30 ppm (or 3% of the measured value) and a temperature-related error of 5 ppm/°C (or 0.5%/°C).

#### 6.4.2.2 $NH_3$ monitoring

For this gas with direct acidifying and indirect greenhouse effect, the approach is the same as that described for $CO_2$: two Seed SENSECAP S-NH3-01 sensors (range: 0–100 ppm, resolution: 0.1 ppm, accuracy: ±5%) were installed in the inlet and outlet air channels. These sensors were coupled with waterproof aviation-grade probes (model SKU 101990862) and communicated using the MODBUS-RTU RS485 protocol.

#### 6.4.2.3 Additional sensors

An additional set of sensors was installed in the diet preparation and larvae separation area, primarily to monitor environmental parameters such as humidity and temperature. These sensors also evaluated air quality for operators by monitoring volatile organic compounds (VOCs) and $CO_2$. A multisensor (Hotcold HCEM-08K) with MODBUS communication was used for this purpose. To assess substrate conditions, three SENSECAP S-Soil MT-02 sensors were installed in the fattening area to measure soil temperature and humidity. These sensors monitored bioconversion progress and potential humidity gradients within breeding trays, both vertically across shelves and horizontally.

6.4.2.4 Set-up of sensors

The sensors were strategically placed within the breeding system to measure specific phenomena. Two pairs of sensors, one for $CO_2$ and one for $NH_3$ were positioned at the entrance of the prototype, specifically in the air inlet tunnel to the heat pump (Figure 1). These sensors measured the baseline concentrations of $CO_2$ and $NH_3$ in the incoming air, providing a reference point for comparing the concentrations resulting from the biotreatment process.

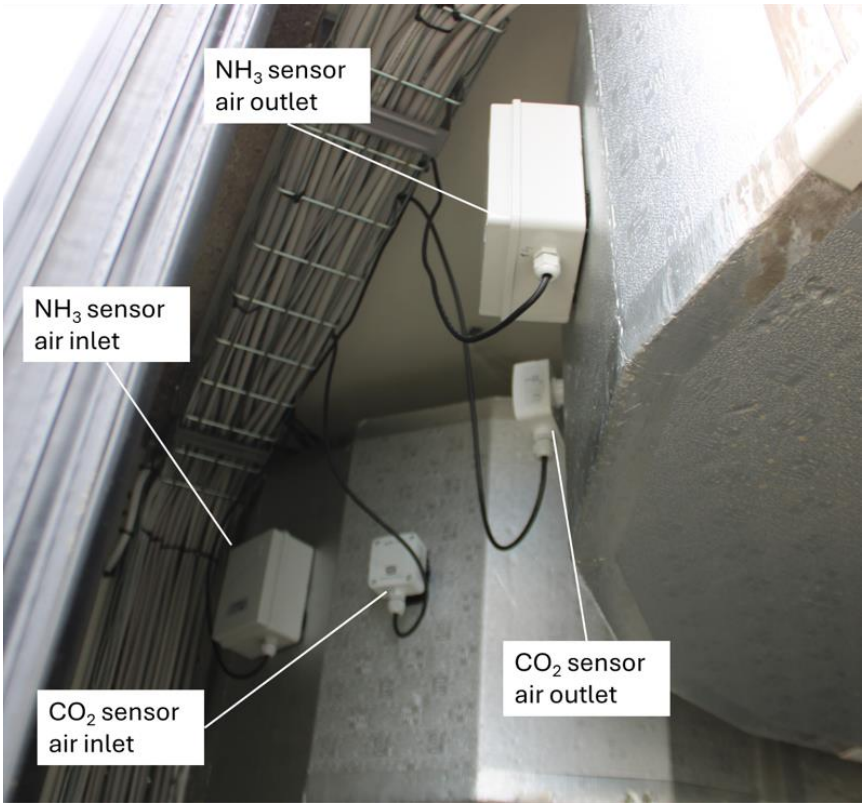



**Figure 6.5.** Location of $CO_2$ and $NH_3$ sensors in the heat pump ventilation ducts.

The second pair of $CO_2$ and $NH_3$ sensors were positioned in the air outlet tunnel downstream of the heat recovery system of the heat pump (Figure 6.5). These sensors were expected to record slightly higher concentrations of both gases due to emissions from larval aerobic metabolism ($CO_2$) and protein degradation ($NH_3$). All four sensors were equipped with gas self-sampling systems, using probes approximately 20 cm long,

positioned in the ducts to avoid turbulent air flows and pressure variations that could interfere with measurements.

The second group of sensors (Figure 6.6) monitored temperature, relative humidity, $CO_2$, and VOC levels within the breeding system. These sensors were strategically placed in the central portion of the container to minimize external environmental disturbances and to control and monitor the environmental conditions set by the thermoregulation and humidification systems. The $CO_2$ sensor, while potentially limited by its measurement range, was included to provide additional insights into the bioconversion process. The VOC sensor was used to assess potential volatile compounds that could pose health risks.

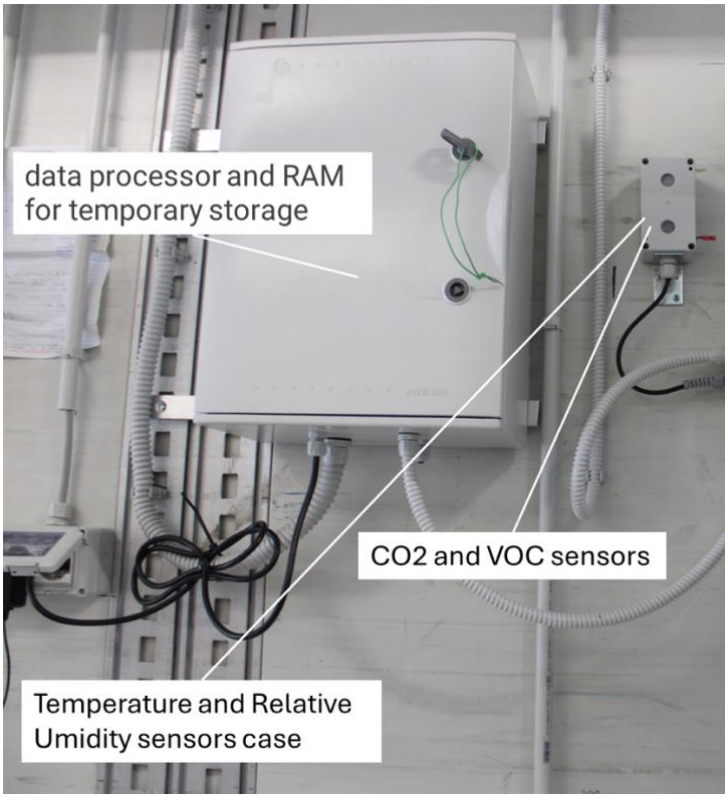



**Figure 6.6.** connection and processing case of all sensors included and measurement system for relative humidity, temperature, VOC, and $CO_2$

The data acquisition system, located near the humidity, temperature, VOC, and $CO_2$ sensors, collects data every 60 seconds and stores it in a 10-minute buffer. This buffered data is then transmitted via SIM card to a cloud-based server for processing and visualization on the ISELQUI technology website.

Additionally, three soil moisture and temperature probes were installed to monitor the conditions within the breeding trays. These sensors were used to track temperature changes resulting from larval activity and to assess the optimal timing for harvesting (Figure 6.7). By monitoring these parameters, it was possible to identify potential issues, such as excessive temperature or dryness, and intervene as needed.



**Figure 6.7.** The three soil moisture and temperature sensors are being tested inside a completely dry soil.

The three soil moisture and temperature sensors were strategically placed near the other sensors in the central area of the prototype, ensuring coverage of the entire fattening area. The sensors were connected to a common junction box with screw connectors (Figure 6.8), with additional junction boxes placed on the opposite side of the prototype to accommodate the length of the sensor cables (5 meters). This configuration enabled comprehensive monitoring of the breeding trays.



**Figure 6.8.** Screw connector block for substrate temperature and humidity sensors.

As mentioned, all data was processed and transmitted via a SIM card-based data transfer system (Figure 6.9). This system, equipped with an antenna, enables data transmission from any location. The initially tested LAN cable connection, although more stable, is much less practical for the conditions of using a breeding system that should work in an agricultural or agro-industrial environment not always equipped with a LAN connection.



**Figure 6.9.** External visual communication system for faults and antenna for data transfer.

The data was stored on a cloud server and made accessible through a web-based platform, "flymonitor," developed by ISELQUI technology. This platform allows for real-time monitoring, data analysis, anomaly detection, sensor comparisons, and data download in various timeframes (1 month, 6 months, 1 year, or the entire recording period) (Figure 6.10).

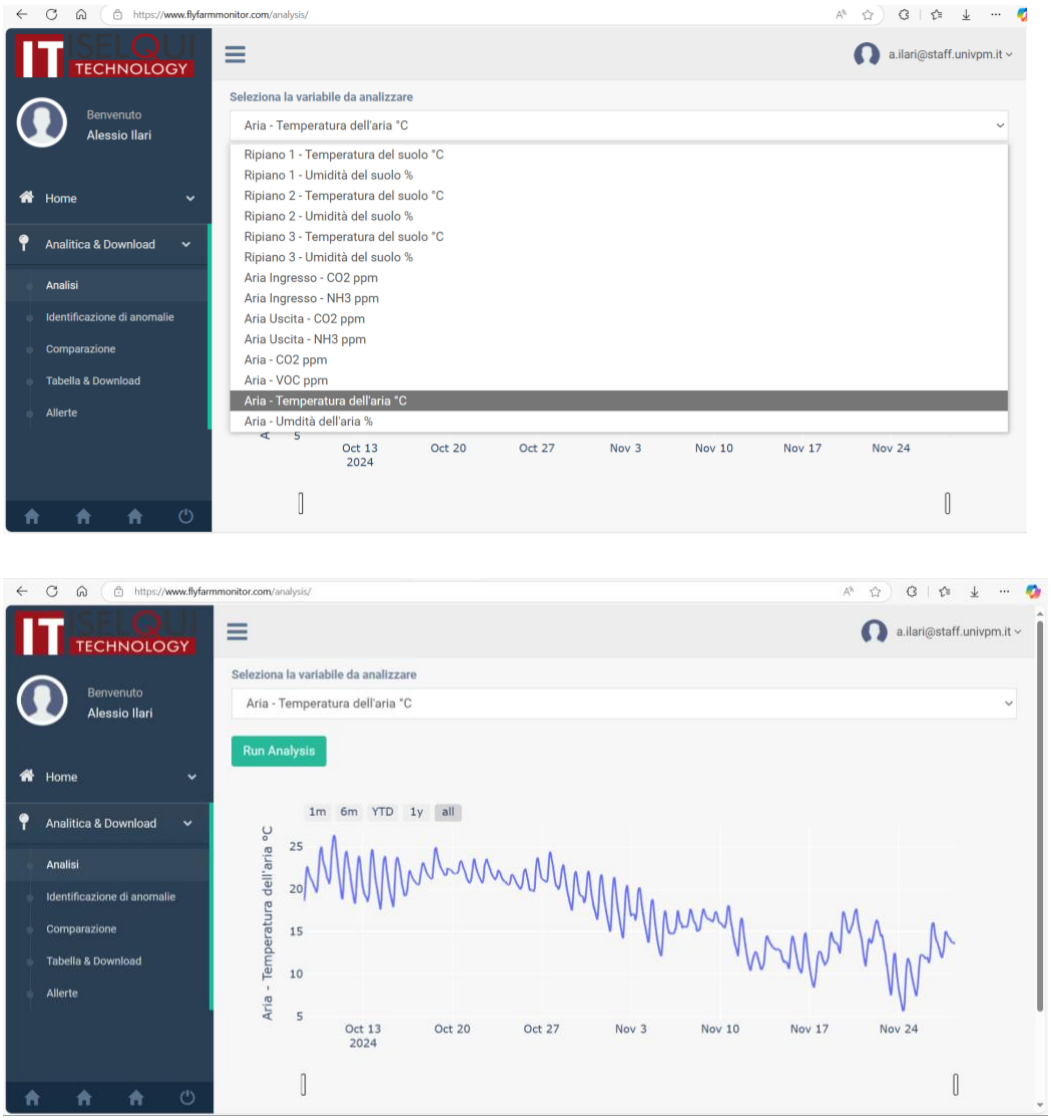



**Figure 6.10.** Two screenshots of the fly monitor show the list of sensors and the ambient temperature data (with the prototype turned off.

*6.4.3 Calculations for larvae growth and substrate reduction*

Larvae growth and substrate reduction performances of insects were computed through the following indices:

i. Bioconversion rate (BCR %) = (total final larvae mass (g)/total substrate mass (g)) * 100
ii. Assimilated feed (g DM) = total substrate dry mass (g) – total frass dry mass (g)
iii. Feed conversion rate (FCR) = (assimilated feed (g))/ (final larvae dry mass (g))
iv. Substrate reduction (%) = (assimilated feed (g)/substrate dry mass(g)) * 100
v. Larval survival (%) = (final larval number/initial larval number) * 100

The initial larval mass was assumed to be zero.

*6.4.4 Life cycle inventory of BSFL breeding*

The experimental data obtained was scaled up to an industrial level for the LCA model construction. The pilot plant, capable of managing 132 trays/cycle, can treat approximately 1.58 t of residue per cycle. Assuming a 10-day bioconversion period and 36 cycles/yr (continuous operation), approximately 57 t of residue can be treated annually, yielding 7.13 t of fresh BSFL biomass (assuming 1.5 kg larvae/tray).

The breeding process comprises three main stages: love-cage and egg hatchery management, bioconversion management, and post-harvest management. The process begins with mating BSF flies and laying the eggs in a love-cage. This cage, of dimensions 140 cm x 130 cm x 100 cm, is constructed of aluminum poles, polycarbonate boards, a polypropylene net, and rubber. It can accommodate 12,500 adult flies per cycle, with half being females. Each female is estimated to lay 300 eggs, with a 50% hatching rate, producing 937,500 larvae/cycle. For this study, the infrastructure of the love-cage was considered, and the lifespan was assumed to be 30 years. Unfortunately, due to elevated temperatures caused by a malfunctioning humidifier, the pilot plant could not produce the required number of larvae. As a result, larvae were sourced from an external supplier – Smart Bug S.r.l (Ponzano Veneto, Italy).

The transportation of residues and larvae to the mass-rearing system was also considered. The fresh residue came from the processing of horticultural products by C.O.VAL. M. - O.R.T.O. Verde S.c.a.p.a. The wheat residue was sourced from a wheat milling company (Italcer Soc. Coop. Agr.), while the chickpeas were obtained from a seed company (Società Produttori Sementi S.p.a.). A 100% load capacity was assumed for residue transportation. Table 6.2 provides details on the transportation distances and modes.

**Table 6.2.** Transportation details on materials to the BSFL biotreatment facility.

| Product | Province | Distance (km) | Transport means |
|---|---|---|---|
| Fresh residue | Senigalia | 48 | Lorry (16 – 32 t, EURO 6) |
| Wheat residue | Osimo | 9.98 | Lorry (16 – 32 t, EURO 6) |
| Chickpea residue | San Serevino | 40.7 | Lorry (16 – 32 t, EURO 6) |
| New larvae | Treviso | 398 | Lorry (3.5 – 7.5 t, EURO 6) |

Concerning infrastructure, the shipping container itself was not considered as it was assumed to be at the end of its life and reused. The materials for internal structures were included, primarily stainless steel support poles, the ventilation system (stainless steel and polyurethane), the scaffold for holding trays (stainless steel), and the polypropylene plastic trays. Regarding equipment, the materials for the milling machine and multidimensional sieve were considered, with lifespans of 10 and 15 years, respectively. Data for the BSFL production at the breeding facility are summarized in Table 6.3.

**Table 6.3.** Inventory analysis of BSFL bioconversion activity. Average data are reported per FU (1 t of fresh larvae produced).

| Parameter | Unit | Amount |
|---|---|---|
| Land use | $m^2$ | 3.43 – 4.14 |
| Fresh residue | t | 4.57 – 6.44 |
| Dry residue | t | 1.64 – 2.44 |
| Fresh residue transport | tkm | 272 - 309 |
| Dry residue transport | tkm | 18 - 99 |
| Larvae transport | tkm | 31 |
| Water (cleaning) | kg | 2.19 – 2.64 |
| Sodium hypochlorite | kg | 0.60 – 0.73 |
| Soap | kg | 0.60 – 0.73 |
| *Electricity* | | |
| Heat pump fans, 0.5kWh per fan (two fans) | kWh | 519 - 627 |

| | | |
|---|---|---|
| Heat pump, 10 kWh | kWh | 1.04E+04 – 1.25E+04 |
| Sieve, 0.5kWh | kWh | 24 – 29 |
| Mill, 22 kWh (100 kg/h) | kWh | 1.42E+03 – 1.78E+03 |
| *Equipment* | | |
| Mill (iron) | kg | 3.53 – 4.35 |
| Mill (polycarbonate) | kg | 0.012 – 0.015 |
| Multidimensional sieve (stainless steel) | kg | 0.12 – 0.15 |
| Multidimensional sieve (concrete) | kg | 0.80 – 0.97 |
| Multidimensional sieve (iron) | kg | 0.28 – 0.33 |
| *Scaffold* | | |
| Scaffold (poles_zinc plated steel) | kg | 0.75 – 0.90 |
| Scaffold (board_zinc plated steel) | kg | 1.73 – 2.09 |
| scaffold holder_zinc plated steel | kg | 0.15 – 0.18 |
| Scaffold (board holder_zinc plated steel) | kg | 0.85 – 1.02 |
| Scaffold (back line_zinc plated steel) | kg | 0.02 – 0.03 |
| Scaffold (back line lock _zinc plated steel) | kg | 7.05E-03 – 8.51E-03 |
| *Tray* | | |
| Plastic tray for bioconversion (polypropylene) | kg | 1.37 – 1.65 |
| *Internal structural support* | | |
| Poles (zinc plated steel) | kg | 0.24 – 0.29 |
| Ventilation pipe for heat distribution (chromium) | kg | 0.17 – 0.21 |
| Ventilation system (polyurethane) | kg | 0.12 – 0.15 |
| *Nursery* | | |
| Polycarbonate board | kg | 0.15 – 0.18 |
| Net (polypropylene) | kg | 3.09E-03 - 3.72E-03 |
| Aluminium poles | kg | 0.02 – 0.03 |
| Rubber | kg | 8.69E-03 – 1.05E-02 |
| *Output* | | |
| Frass (fresh weight) | t | 1.13 – 1.47 |
| $N_2O$ | kg | 8* |
| $CH_4$ | g | 18.8* |

*Secondary data from Ermolaev et al. [91].

*6.4.5 Allocation and substitutions*

Currently, there is no standardized market for BSFL, making it challenging to establish an economic allocation between the larvae and the frass. The frass can be regarded as a substitute for manure, which generally has very low economic value or even a negative value when producers must pay for its disposal. As a result, all environmental impacts were allocated to the fresh BSFL biomass.

For the potential use of BSFL as an alternative protein source in animal feed, BSFL was assumed to replace conventional protein sources such as soybean meal, fishmeal, or pea protein. The selected processes for soybean meal, fishmeal, and pea protein in the

Ecoinvent database were adjusted to reflect their respective protein contents, ensuring an accurate comparison with the total protein content of BSFL. The protein content and the comparison with conventional protein sources are detailed in Table 6.4.

Additionally, BSFL fat was evaluated as a potential feedstock for biodiesel production, potentially replacing vegetable oils like rapeseed oil, palm oil, soybean oil, and sunflower oil. The total lipid content and higher heating value (HHV) of BSFL lipids were compared with those of the selected vegetable oils (Table 6.4). To ensure fair comparison and reduce uncertainty, the biodiesel production process was excluded from the system boundaries, focusing only on the phases up to seed production.

**Table 6.4.** Protein, fat, and energy content of BSFL and conventional sources.

| | **Moisture content (%)** | **Protein (% DM)** | **Fat (% DM)** | **HHV (MJ/kg DM)** |
|---|---|---|---|---|
| BSFL | 68 [1] | 39 [2] | 40 [2] | 37.9 [3] |
| *Protein sources* | | | | |
| Soybean meal | 11 [4] | 45 [4] | - | - |
| Fishmeal | 10 [5] | 64 [6] | - | - |
| Pea protein | 12 [7] | 21 [7] | - | - |
| *Lipid sources* | | | | |
| Rapeseed | 10 [8] | - | 48 [8] | 41.1 [9] |
| Palm fruit | 24 [10] | - | 77 [10] | 40.6 [9] |
| Soybean | 12 [8] | - | 21[11] | 39.7 [9] |
| Sunflower | 15 [12] | - | 50 [13] | 40.6 [9] |

Sources: [1] – Own study, [2] – [226], [3] – [227], [4] – [228], [5] – [32], [6] – The Ecoinvent process for *fishmeal production*, [7] – [229], [8] – [230], [9] – [231], [10] – [232], [11] – [233], [12] – [234], [13] –[235].

## 6.5 RESULTS AND DISCUSSION

### *6.5.1 Growth and bioconversion of BSFL on different diets*

Tables 6.5 and 6.6 present data and results on the larval and substrate characteristics, as well as the larval growth and substrate reduction performance across various diets. The average fresh larval weight per tray was approximately 1.5 kg, with Diet 5 showing the highest performance, having an average yield of 1.75 kg per tray. In contrast, Diet 6 showed the poorest performance, attributed to elevated temperatures caused by a malfunctioning humidifier. This issue led to significant larval mortality and escape, accounting for the low yield obtained. Once the environmental conditions were stabilized

by deactivating the humidifier, Diet 5 demonstrated better results. These findings emphasize the importance of maintaining optimal temperature control for successful larval development. Additionally, substrates with lower lignin and lignocellulose content were easier to process and separate after harvest, further contributing to improved efficiency in the rearing system.

Regarding Diet 5, the disposal of liquid digestate poses significant challenges for biogas producers due to strict regulations on application amounts and seasonal limitations. The instability of liquid digestate, primarily due to its ammonia content, necessitates large storage tanks or lakes, which are costly and environmentally problematic. Open storage often releases methane, ammonia, and nitrous oxide emissions [236, 237]. Therefore, using liquid digestate as a substrate for BSFL production offers the dual benefit of reducing storage and emission issues while alleviating competition between agro-residues for biogas production.

The dry matter content of the BSFL was approximately 33%, consistent with other studies [111, 238]. The bioconversion rate ranged from 12% to 16%, aligning with previous findings on BSFL reared on various substrates. Similarly, the feed conversion ratio was between 4 and 6, which was also comparable to results reported in other studies [37, 239]. The substrate reduction ranged between 73 – 81% within 10 days of treatment, slightly higher than values reported for fruit and vegetable waste diets in other studies [240, 241]. However, Lalander et al. [242] reported lower substrate reduction capacities of (50 – 60%) for fruits and vegetables, which may be attributed to the higher lignin content of these materials, as lignin-rich substrates are generally less efficiently broken down in the rapid BSFL composting process.

While several studies have highlighted the influence of substrate composition on BSFL development, growth, and waste reduction efficiency, further detailed analysis of the diets, larvae, and frass was not feasible due to several constraints. However, this study demonstrates the suitability of agro-industrial residues from frozen vegetable production and liquid digestate as substrates for BSFL. Nonetheless, careful consideration of microbial contamination risks is essential for safe and sustainable BSFL production.

**Table 6.5.** Larvae and substrate biomass characteristics per tray (12 kg of fresh substrate).

| Substrate | Final larvae weight average (kg) | Frass fresh matter (kg) | Larvae dry matter (%) | Larvae dry weight (kg) | Frass dry matter (%) | Frass dry mass (kg) |
|---|---|---|---|---|---|---|
| Diet 1 (tomato+ wheat) | 1.49 | 2.22 | 32 | 0.48 | 43 | 0.95 |
| Diet 2 (onion + wheat) | 1.45 | 2.13 | 32 | 0.46 | 38 | 0.81 |
| Diet 3 (peas + chickpea) | 1.63 | 1.99 | 34 | 0.55 | 44 | 0.87 |
| Diet 4 (tomato + chickpea) | 1.48 | 1.68 | 33 | 0.49 | 40 | 0.67 |
| Diet 5 (liquid digestate + wheat) | 1.75 | 2.40 | 34 | 0.60 | 39 | 0.94 |
| Diet 6 (peas + wheat) | 0.67 | 3.14 | 32 | 0.22 | 40 | 1.26 |

**Table 6.6.** Larvae growth and substrate reduction performance indices.

| Substrate | Bioconversion rate (%) | Assimilated feed (g DM) | Feed conversion rate | Substrate reduction (%) |
|---|---|---|---|---|
| Diet 1 (tomato+ wheat) | 12 | 2.65 | 5.54 | 73 |
| Diet 2 (onion + wheat) | 12 | 2.79 | 6.02 | 78 |
| Diet 3 (peas + chickpea) | 14 | 2.73 | 4.93 | 76 |
| Diet 4 (tomato + chickpea) | 12 | 2.93 | 5.98 | 81 |
| Diet 5 (liquid digestate + wheat) | 15 | 2.66 | 4.47 | 74 |
| Diet 6 (peas + wheat) | 6 | 2.34 | 10.90 | 65 |

Larvae growth and substrate reduction performances of insects were computed through the following indices:

1. Bioconversion rate (BCR; %) = (total final larvae mass (g)/total substrate mass (g)) * 100
2. Assimilated feed (g DM) = total substrate dry mass (g) – total frass dry mass (g)
3. Feed conversion rate (FCR) = (assimilated feed (g))/(final larvae dry mass (g))
4. Substrate reduction (%) = (assimilated feed (g)/substrate dry mass(g)) * 100

**Table 6.7.** Summary of measured parameters by the sensors. Mean, maximum, and minimum values reported for trials.

| Trial period | ST (1) | SM (1) | ST (2) | SM (2) | ST (3) | SM (3) | AirIn $CO_2$ | AirIn $NH_3$ | AirOut $CO_2$ | AirOut $NH_3$ | Air $CO_2$ | AirVOC | AirTemp |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 26/06-5/07 av | 38.75 | 35.90 | 40.78 | 45.043 | 38.61 | 23.48 | 492.1 | 2.80 | 430.4 | 3.01 | 555.74 | 0.19 | 32.46 |
| 26/06-5/07 max | 50.87 | 78.19 | 58.24 | 100 | 53.03 | 87.96 | 696.6 | 16.9 | 552.8 | 15.42 | 879 | 1.85 | 38.8 |
| 26/06-5/07 min | 28.38 | 0 | 27.82 | 0 | 26.78 | 0 | 397.4 | 0 | 397.4 | 0 | 410 | 0.02 | 29.3 |
| 31/07-12/08 av | 38.68 | 34.78 | 39.64 | 48.31 | 40.56 | 36.21 | 549.2 | 4.60 | 439.0 | 5.07 | 684.12 | 0.43 | 32.80 |
| 31/07-12/08 max | 46.42 | 96.6 | 45.96 | 100 | 49.53 | 99.93 | 775.6 | 19.6 | 538 | 19.02 | 1202 | 3.25 | 39.3 |
| 31/07-12/08 min | 28.57 | 0 | 28.09 | 0 | 27.57 | 0 | 397.4 | 0 | 397.4 | 0 | 399 | 0.02 | 28.6 |
| 14/08-28/08 av | 36.35 | 52.18 | 35.80 | 52.99 | 36.23 | 52.48 | 470.3 | 1.89 | 442.9 | 2.59 | 557.32 | 0.06 | 31.60 |
| 14/08-28/08 max | 49.96 | 100 | 53.28 | 100 | 51.46 | 100 | 634 | 21.4 | 572.4 | 19.99 | 923 | 0.27 | 35.3 |
| 14/08-28/08 min | 27.33 | 0 | 27.33 | 0 | 27.4 | 0 | 397.4 | 0 | 397.6 | 0 | 399 | 0.02 | 27.9 |
| 28/08-5/09 av | 29.64 | 4.52 | 29.80 | 4.11 | 29.68 | 4.52 | 428.9 | 0.34 | 435.7 | 0.58 | 509.1 | 0.02 | 30.85 |
| 28/08-5/09 max | 33.23 | 100 | 37.32 | 100 | 33.28 | 100 | 591.2 | 9.49 | 673.6 | 10.4 | 804 | 0.02 | 34.5 |
| 28/08-5/09 min | 26.22 | 0 | 26.22 | 0 | 26.29 | 0 | 397.4 | 0 | 397.6 | 0 | 441 | 0.02 | 27.2 |

ST – Soil temperature, SM – soil moisture, VOC- volatile organic compounds, Temp – temperature, av – average, max – maximum, min – minimum

*6.5.2 Ambient conditions monitoring*

Several phenomena emerged from the analysis of sensor measurements in the BSFL bioconversion prototype, providing both useful insights and critical considerations for system optimization (Table 6.7). The $CO_2$ sensors, used to measure input and output concentrations, recorded values consistent with expected atmospheric concentrations. However, an unexpected trend in which input sensor readings were consistently higher than output readings was observed. This result is counterintuitive, as aerobic bioconversion processes should generate additional $CO_2$.

Three plausible explanations arise for this anomaly. First, interference from nearby biogas plant operations, including emissions from the feedstock, cogeneration unit, flaring torch, and liquid digestate tanks, may have affected sensor readings. Although such disturbances would likely impact both sensors equally, localized variations could account for the discrepancy. Second, the sensitivity limitations of the low-cost sensors may have contributed to measurement inaccuracies. Atmospheric-level gas measurement is particularly challenging with such sensors, and the resulting error could exceed the actual differences in $CO_2$ concentration. Third, disturbances from the prototype's ventilation system, which circulates over 720 $m^3$ of air per hour, likely altered air pressure conditions critical to sensor performance. This issue, combined with sensor limitations, is the most probable cause of the observed discrepancies.

For $NH_3$, sensor measurements during the testing period indicated values fluctuating between zero (in the absence of larvae) and peaks of 10–20 ppm on the third to fourth and seventh to eighth days of the process. Average values remained below 5 ppm, well within the exposure limits established by EU Directive 2019/1831/EU. However, an anomalous pattern similar to $CO_2$ measurements was observed, with higher $NH_3$ concentrations at the input rather than the output. The likely causes include disturbances from digestate storage at the biogas plant and nitrogenous emissions from larvae physiology, which prioritize nitrogenous waste over $CO_2$ production.

The temperature and humidity sensors within the prototype provided additional insights. Substrate temperatures ranged between 30°C and 40°C, generally optimal for bioconversion. However, bacterial fermentation occasionally caused localized spikes above 50 °C, leading to larval migration or death. Humidity trends were consistent with BSFL activity, starting at 4–50% and reducing over time, depending on evaporation periods before harvest. Maintaining optimal humidity is essential. Excessively high humidity impedes substrate evaporation, causing overheating that disrupts the

bioconversion process. Conversely, controlled humidity around 50–70% allowed effective substrate drying and frass separation, which is critical for process efficiency.

Initial tests revealed operational issues with the system's humidifier, which malfunctioned and was subsequently turned off. This change inadvertently resolved high-humidity-related issues, such as larval mortality and escape, demonstrating that high atmospheric humidity is unnecessary for larval rearing. Nighttime temperatures, often reaching 40°C, coupled with the removal of excess humidity, supported stable bioconversion conditions, enabling larvae to thrive.

In conclusion, although the prototype highlighted critical challenges such as sensor limitations and environmental control issues, these findings inform potential optimizations. Addressing sensor sensitivity, improving ventilation design, and maintaining balanced temperature and humidity conditions are crucial for scaling BSFL bioconversion systems effectively.

### *6.5.3 Comparison of insect diets (FU: 1 t fresh BSFL)*

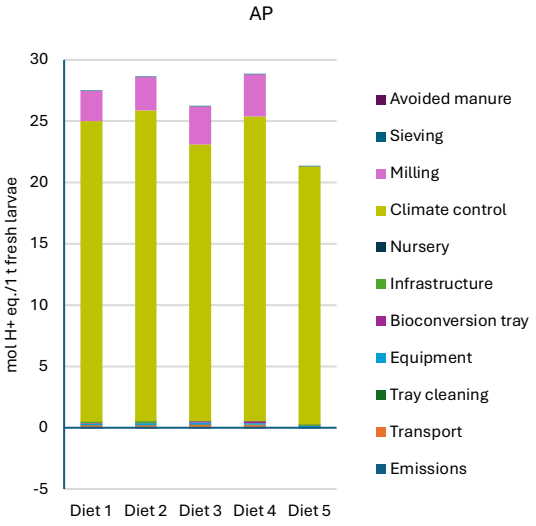
This section discusses the environmental impacts of BSFL production on different diets. Table 6.8 presents the absolute impact scores for assessed midpoint impact categories, while Figure 6.5 illustrates the contributions of various inputs and emissions to the overall impact scores of relevant environmental indicators.

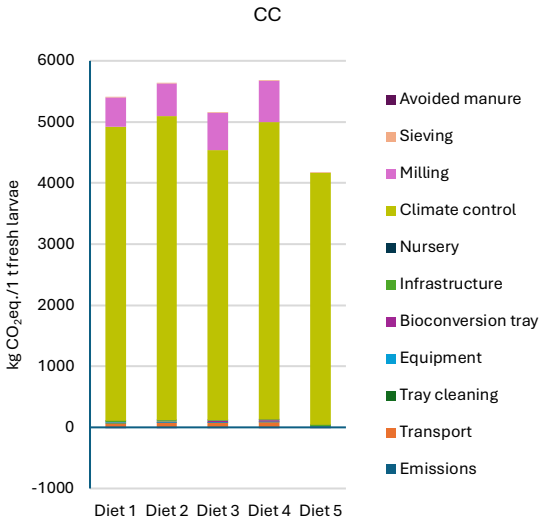
The results indicate no substantial difference in environmental performance among the assessed diets. Nonetheless, Diet 5 showed the best performance, while the others showed similar results. The performance was primarily influenced by feed substrate, larval, and frass yield. Diet 5 benefited from the absence of milling, as both liquid digestate and milled wheat residue were used without further processing. Additionally, the on-site availability of liquid digestate eliminated the need for residue transportation. The highest larval yield was observed for Diet 5. However, it is important to note that Diet 6 was excluded from the analysis due to challenges with the humidifier during the breeding process, which compromised the data quality.

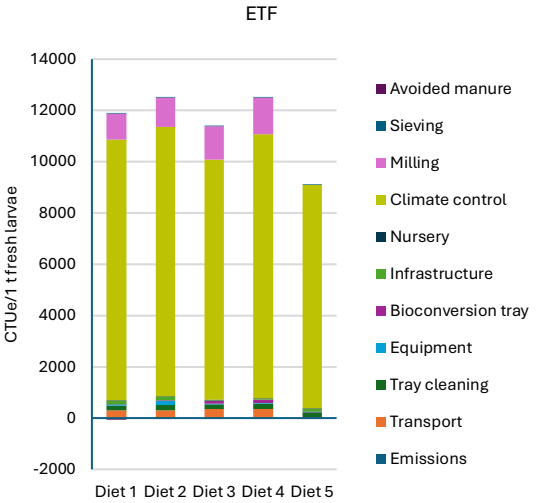
**Table 6.8.** Midpoint impact assessment scores for BSFL production on different diets. (Scores reported per 1 tonne fresh larvae).

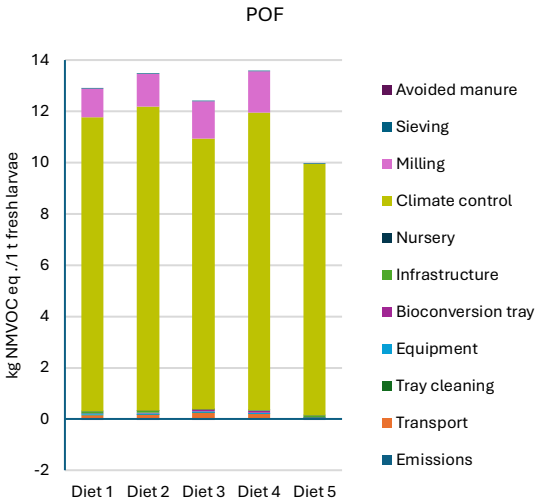
| Impact | Unit | Diet 1 | Diet 2 | Diet 3 | Diet 4 | Diet 5 |
|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 2.75E+01 | 2.86E+01 | 2.63E+01 | 2.89E+01 | 2.13E+01 |
| CC | kg $CO_2$ eq. | 5.40E+03 | 5.63E+03 | 5.16E+03 | 5.68E+03 | 4.17E+03 |
| ETF | CTUe | 1.19E+04 | 1.24E+04 | 1.14E+04 | 1.25E+04 | 9.07E+03 |
| PM | disease inc. | 1.26E-04 | 1.32E-04 | 1.21E-04 | 1.33E-04 | 9.49E-05 |
| MEU | kg N eq. | 4.37E+00 | 4.56E+00 | 4.20E+00 | 4.60E+00 | 3.38E+00 |
| FEU | kg P eq. | 1.66E+00 | 1.74E+00 | 1.59E+00 | 1.75E+00 | 1.30E+00 |
| TEU | mol N eq. | 4.57E+01 | 4.76E+01 | 4.39E+01 | 4.80E+01 | 3.53E+01 |
| HT_c | CTUh | 6.80E-07 | 7.08E-07 | 6.47E-07 | 7.13E-07 | 5.10E-07 |
| HT_nc | CTUh | 2.17E-06 | 2.27E-06 | 2.08E-06 | 2.30E-06 | 1.67E-06 |
| IR | kBq U-235 eq. | 7.66E+02 | 7.99E+02 | 7.30E+02 | 8.04E+02 | 5.94E+02 |
| LU | Pt | 3.44E+04 | 3.59E+04 | 3.29E+04 | 3.62E+04 | 2.65E+04 |
| ODP | kg CFC11 eq. | 8.40E-04 | 8.76E-04 | 8.02E-04 | 8.83E-04 | 6.46E-04 |
| POF | kg NMVOC eq. | 1.29E+01 | 1.34E+01 | 1.24E+01 | 1.36E+01 | 9.93E+00 |
| FRD | MJ | 8.45E+04 | 8.82E+04 | 8.07E+04 | 8.89E+04 | 6.53E+04 |
| MRD | kg Sb eq. | 5.30E-02 | 5.52E-02 | 5.05E-02 | 5.56E-02 | 4.12E-02 |
| WU | $m^3$ depriv. | 4.82E+03 | 5.03E+03 | 4.60E+03 | 5.06E+03 | 3.76E+03 |

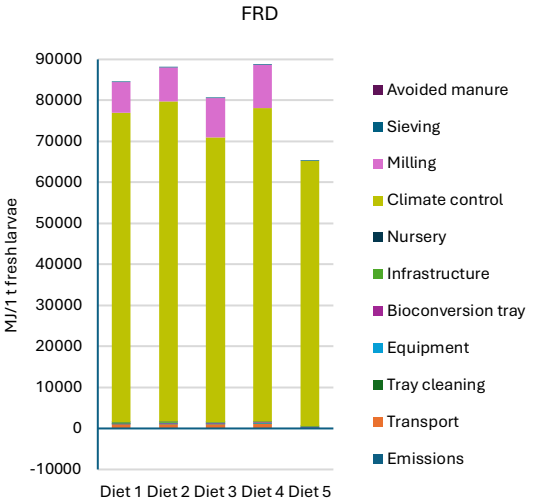
(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

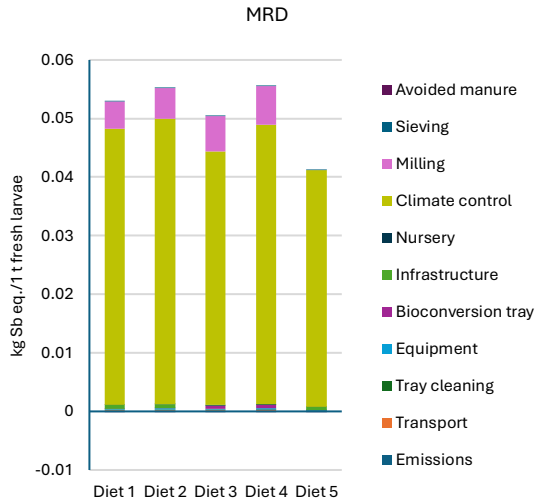

AP
mol H+ eq./1 t fresh larvae
30
25
20
15
10
5
0
-5
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

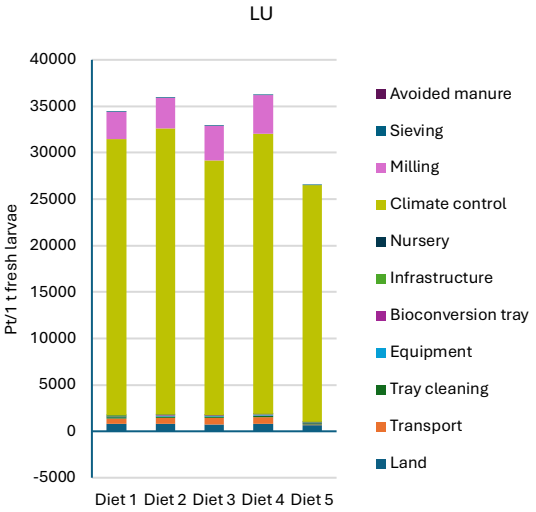

CC
kg CO2eq./1 t fresh larvae
6000
5000
4000
3000
2000
1000
0
-1000
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

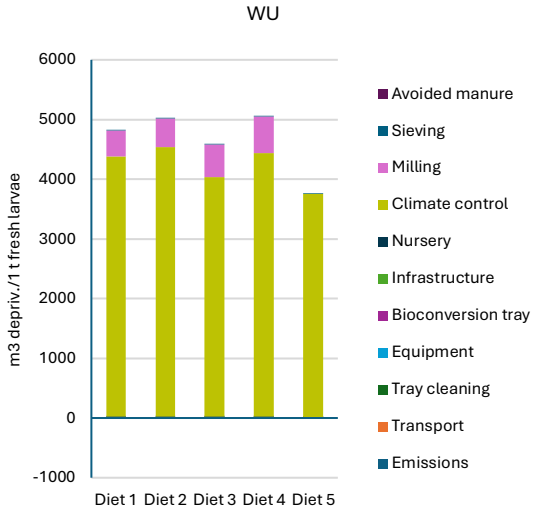

ETF
CTUe/1 t fresh larvae
14000
12000
10000
8000
6000
4000
2000
0
-2000
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

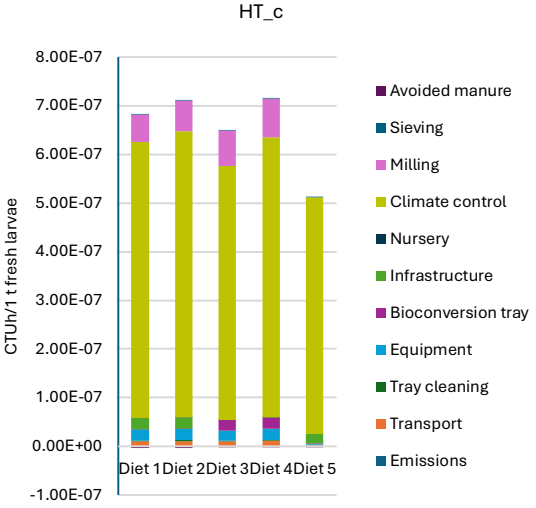

POF
kg NMVOC eq./1 t fresh larvae
14
12
10
8
6
4
2
0
-2
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

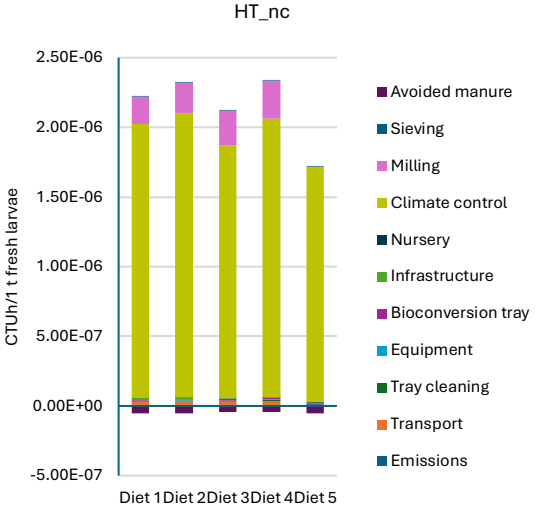

FRD
MJ/1 t fresh larvae
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
-10000
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

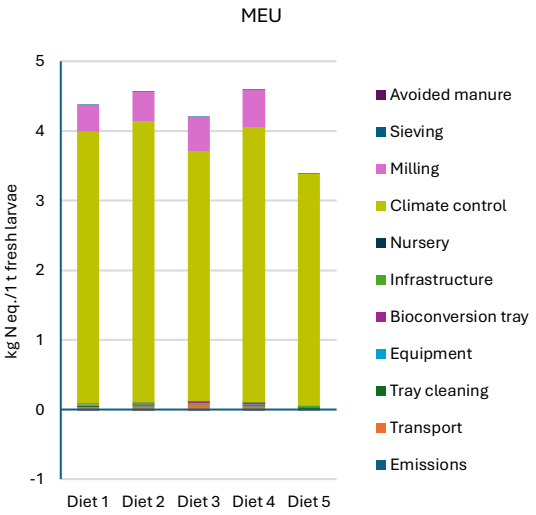

MRD
kg Sb eq./1 t fresh larvae
0.06
0.05
0.04
0.03
0.02
0.01
0
-0.01
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

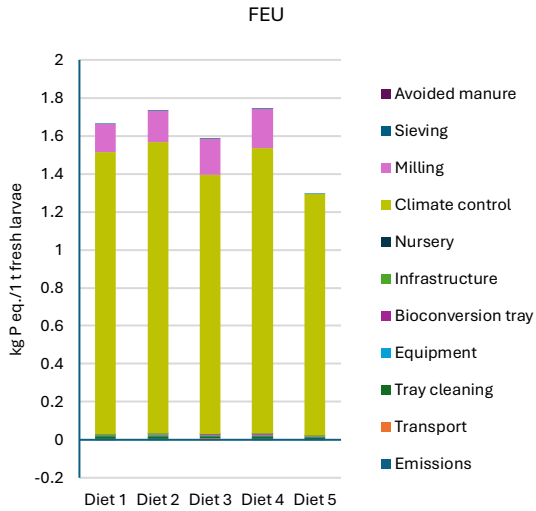

LU
Pt/1 t fresh larvae
40000
35000
30000
25000
20000
15000
10000
5000
0
-5000
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Land


WU
m3 depriv./1 t fresh larvae
6000
5000
4000
3000
2000
1000
0
-1000
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions


HT_c
CTUh/1 t fresh larvae
8.00E-07
7.00E-07
6.00E-07
5.00E-07
4.00E-07
3.00E-07
2.00E-07
1.00E-07
0.00E+00
-1.00E-07
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions


HT_nc
CTUh/1 t fresh larvae
2.50E-06
2.00E-06
1.50E-06
1.00E-06
5.00E-07
0.00E+00
-5.00E-07
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions


MEU
kg N eq./1 t fresh larvae
5
4
3
2
1
0
-1
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions


FEU
kg P eq./1 t fresh larvae
2
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
-0.2
Diet 1 Diet 2 Diet 3 Diet 4 Diet 5
Avoided manure
Sieving
Milling
Climate control
Nursery
Infrastructure
Bioconversion tray
Equipment
Tray cleaning
Transport
Emissions

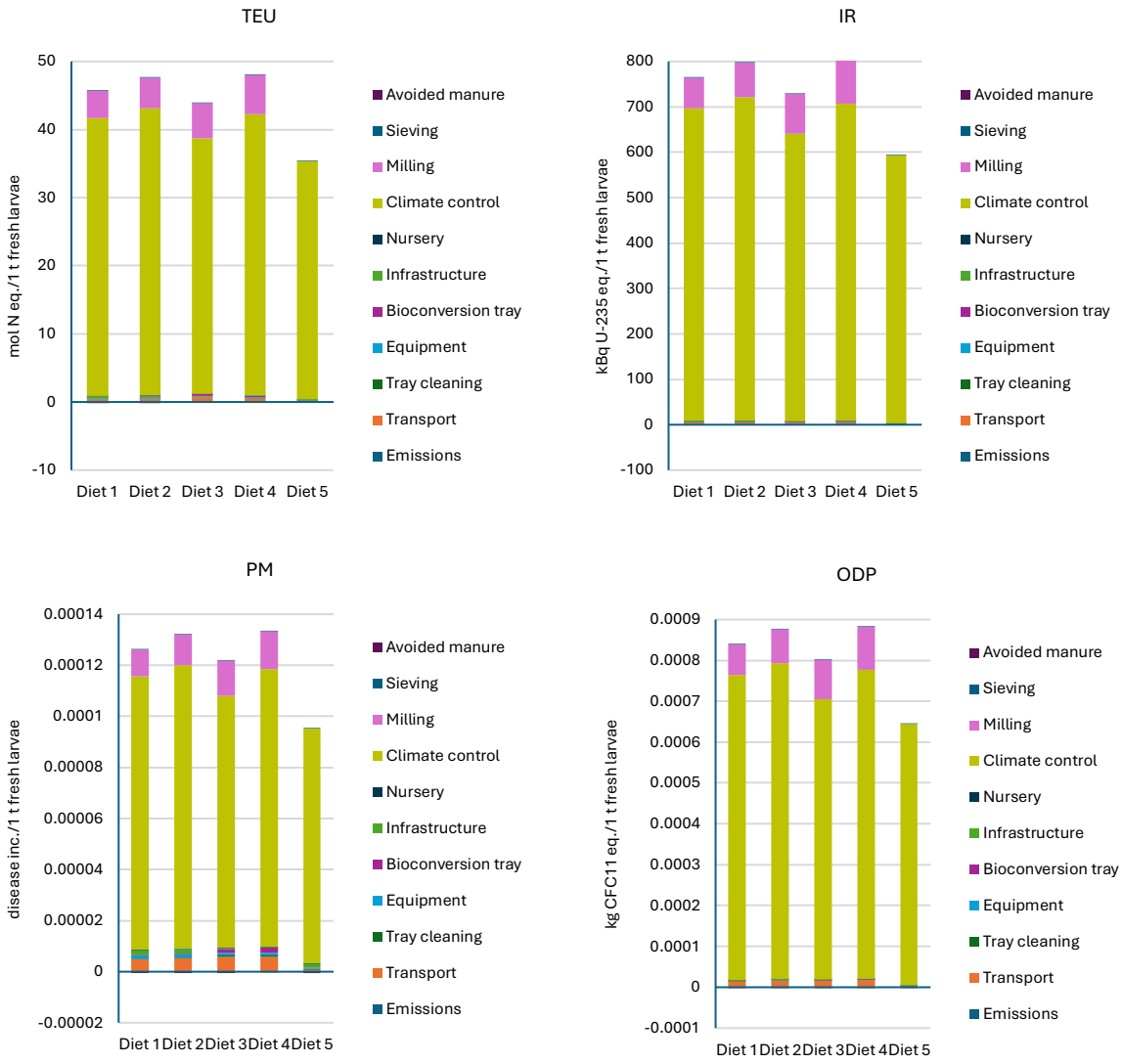



**Figure 6.11.** Contribution analysis for BSFL production on different diets. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

From the contribution analysis (Figure 6.11), electricity consumption for climate control was the primary driver of environmental impact, contributing between 80% and 95% to the total impact across all assessed categories. The initial plan was to utilize residual heat from the biogas plant for climate control in the prototype. However, technical and authorization challenges prevented its implementation. As an alternative, an electric heat pump was installed for climate control, with electricity sourced from the biogas plant. Although this electricity is considered renewable, the heat pump's high energy consumption significantly contributed to the overall impact. This suggests that

the electric heat pump may not be the most suitable option for the prototype. Opting for an air conditioning system with lower energy requirements could potentially improve energy efficiency and reduce the environmental impact associated with climate control.

Similar studies have identified energy use as a major hotspot in BSFL biotreatment or production [27, 63, 64, 243]. While climate control is crucial in colder climates, ambient conditions may be sufficient to ensure BSFL productivity in warmer regions. In temperate zones, heat pumps can be switched off during warmer months to conserve energy. The impact on electricity production is also country-specific, as countries with a greater share of renewable energy in the average country mix will have a lower impact on some impact categories like CC. Therefore, the geographical location and seasonality of BSFL production can significantly influence the overall environmental performance.

Other inputs had relatively lower impacts. For instance, milling accounted for approximately 9% of the total impacts across all assessed categories, while transportation contributed less than 4%. The minimal impact of transportation highlights the importance of local sourcing of raw materials combined with an efficient logistics system. This approach not only provides environmental benefits but also offers significant economic advantages [244]. The avoided product of manure provided a credit of less than 1% across most impact categories, except for HTC_nc, where it contributed 2.3%. Manure, an organic by-product of animal production, has limited economic value, with most environmental impacts attributed to meat production rather than the manure itself. Thus, not much credit was obtained from its avoided production. However, this analysis did not include emissions from manure application to soil, as similar emissions were not assessed for the frass obtained in the study.

### *6.5.4 BSFL vs. convention protein sources (FU: provision of 1 kg of protein in feed)*

Beyond waste disposal, BSFL can replace critical feedstuff components like fishmeal, soybean meal, and pea protein. To assess their environmental performance, a FU of 1 kg of protein was used to compare BSFL-based diets (without avoided manure) with these conventional protein sources (Table 6.9). The results indicate that, although all BSFL diets performed relatively poorly across most impact categories, they outperformed soybean meal and pea protein in terms of land use (LU). The significant environmental impact of BSFL production is primarily attributed to electricity consumption, which accounts for over 85% of the total impact. Optimizing energy efficiency or adopting alternative, cleaner energy sources could significantly improve the overall

environmental performance of BSFL production and make it more comparable to conventional animal feed protein sources.

Currently, using BSFL fed on digestate (a form of manure) as livestock feed is prohibited. However, given the promising growth performance of BSFL on liquid digestate and wheat substrates and considering the natural feeding behavior of some livestock such as chickens, exploring this option could offer a sustainable and circular approach to food production. Fishmeal had the lowest environmental burden among the protein sources assessed. However, its production can have significant environmental impacts, including biodiversity loss, species depletion, and microplastic pollution, which are not fully captured presently in LCAs [245, 246]. Additionally, increasing soybean and pea protein production to meet growing demand would require expansion and intensification of crop cultivation, leading to increased resource use (land, water, agrochemicals, etc.) and emissions and their related negative environmental impacts. Therefore, BSFL presents a viable and sustainable alternative to these conventional protein sources for animal feed.

*6.5.5 BSFL vs. conventional vegetable oil sources for producing biodiesel (FU: 1 MJ biodiesel energy)*

BSFL fat has the potential to serve as a feedstock for biodiesel production, providing an alternative to traditional sources like rapeseed, palm, soybean, and sunflower [247, 248]. Based on fat content and higher heating value (HHV), the environmental impacts of producing 1 MJ of biodiesel from BSFL-derived fat were compared to conventional sources (Table 6.10). The results show that the environmental burden of using BSFL (without avoided manure) as a biodiesel source is substantially higher than that of vegetable oils across all impact categories except for LU. This finding aligns with the results observed for using BSFL protein in feed production. The CC impact of BSFL for energy production was two orders of magnitude higher than that of rapeseed, palm fruit, and sunflower oils. However, except for palm fruit, BSFL had lower LU scores than the other feedstocks. The lower LU impact of BSFL can be attributed to its efficient conversion of organic waste into biomass, with less land requirement. However, the high energy consumption associated with BSFL production, mainly for climate control, significantly contributes to its overall environmental performance. Addressing this issue, such as through energy efficiency improvements or renewable energy sources, could substantially enhance the sustainability of BSFL-based biodiesel production.

Although biodiesel produced from vegetable oils can be more expensive than petroleum diesel and may compete with food production, BSFL fat, mainly when derived from organic waste

streams like liquid digestate or manure, offers a more sustainable and potentially cost-effective alternative [248, 249]. Extensive production of bioenergy from dedicated crops could possibly lead to land use changes, both direct and indirect, with possible economic, social, and environmental implications [250, 251]. However, by addressing the energy consumption challenges associated with BSFL production, this approach can contribute to a more sustainable bioenergy future, reducing the need for land-intensive bioenergy crops.

**Table 6.9.** Environmental impact of different diets compared to soybean, fishmeal and pea protein per 1 kg protein on dry matter basis.

| Impact | Unit | Soybean meal | Fishmeal | Pea protein | Diet 1 | Diet 2 | Diet 3 | Diet 4 | Diet 5 |
|---|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 7.33E-03 | 2.52E-02 | 1.34E-02 | 6.86E-02 | 7.15E-02 | 6.56E-02 | 7.21E-02 | 5.32E-02 |
| CC | kg $CO_2$ eq. | 6.57 | 1.93 | 1.90 | 13.49 | 14.07 | 12.89 | 14.19 | 10.42 |
| ETF | CTUe | 24.58 | 9.96 | 62.05 | 29.50 | 30.78 | 28.28 | 31.12 | 22.56 |
| PM | disease inc. | 6.83E-08 | 2.50E-07 | 8.32E-08 | 3.14E-07 | 3.28E-07 | 3.03E-07 | 3.32E-07 | 2.36E-07 |
| MEU | kg N eq. | 1.40E-02 | 5.80E-03 | 6.01E-02 | 1.09E-02 | 1.14E-02 | 1.05E-02 | 1.15E-02 | 8.43E-03 |
| FEU | kg P eq. | 5.03E-04 | 1.34E-03 | 7.38E-04 | 4.16E-03 | 4.34E-03 | 3.97E-03 | 4.37E-03 | 3.24E-03 |
| TEU | mol N eq. | 2.56E-02 | 5.93E-02 | 4.49E-02 | 1.14E-01 | 1.19E-01 | 1.09E-01 | 1.20E-01 | 8.80E-02 |
| HT_c | CTUh | 1.84E-09 | 6.38E-10 | 7.53E-10 | 1.69E-09 | 1.76E-09 | 1.61E-09 | 1.78E-09 | 1.27E-09 |
| HT_nc | CTUh | 3.20E-09 | 9.50E-10 | 4.57E-09 | 5.29E-09 | 5.53E-09 | 5.09E-09 | 5.64E-09 | 4.05E-09 |
| IR | kBq U-235 eq. | 4.31E-02 | 9.55E-02 | 9.99E-02 | 1.92E+00 | 2.00E+00 | 1.83E+00 | 2.01E+00 | 1.49E+00 |
| LU | Pt | 261.25 | 13.33 | 744.80 | 85.93 | 89.63 | 82.24 | 90.40 | 66.18 |
| ODP | kg CFC11 eq. | 1.30E-07 | 1.63E-06 | 2.39E-07 | 2.10E-06 | 2.19E-06 | 2.00E-06 | 2.21E-06 | 1.61E-06 |
| POF | kg NMVOC eq. | 7.63E-03 | 1.58E-02 | 1.20E-02 | 3.21E-02 | 3.35E-02 | 3.09E-02 | 3.38E-02 | 2.48E-02 |
| FRD | MJ | 9.46 | 21.33 | 17.32 | 211.16 | 220.20 | 201.57 | 222.03 | 163.11 |
| MRD | kg Sb eq. | 5.56E-06 | 1.29E-05 | 1.89E-05 | 1.32E-04 | 1.38E-04 | 1.26E-04 | 1.39E-04 | 1.03E-04 |
| WU | $m^3$ depriv. | 0.25 | 0.91 | 16.53 | 12.05 | 12.56 | 11.49 | 12.65 | 9.40 |

(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 6.10.** Environmental impact of different diets compared to rapeseed, palm fruit, sunflower, and soybean per 1 MJ of fat energy on dry matter basis.

| Impact | Unit | Rapeseed | Palm | Sunflower | Soybean | Diet 1 | Diet 2 | Diet 3 | Diet 4 | Diet 5 |
|---|---|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 1.32E-03 | 1.01E-04 | 8.80E-04 | 3.65E-04 | 2.01E-03 | 1.89E-03 | 1.73E-03 | 1.90E-03 | 1.40E-03 |
| CC | kg $CO_2$ eq. | 0.05 | 0.02 | 0.07 | 0.48 | 0.40 | 0.37 | 0.34 | 0.37 | 0.27 |
| ETF | CTUe | 1.31 | 0.60 | 1.47 | 1.54 | 0.86 | 0.81 | 0.75 | 0.82 | 0.60 |
| PM | disease inc. | 9.22E-09 | 8.40E-10 | 5.73E-09 | 3.06E-09 | 9.21E-09 | 8.65E-09 | 7.98E-09 | 8.76E-09 | 6.22E-09 |
| MEU | kg N eq. | 7.95E-04 | 8.82E-05 | 1.74E-03 | 1.05E-03 | 3.20E-04 | 3.00E-04 | 2.76E-04 | 3.03E-04 | 2.23E-04 |
| FEU | kg P eq. | 1.55E-05 | 1.51E-06 | 1.72E-05 | 3.30E-05 | 1.22E-04 | 1.14E-04 | 1.05E-04 | 1.15E-04 | 8.54E-05 |
| TEU | mol N eq. | 5.60E-03 | 4.23E-04 | 3.69E-03 | 1.41E-03 | 3.34E-03 | 3.13E-03 | 2.89E-03 | 3.16E-03 | 2.32E-03 |
| HT_c | CTUh | 1.30E-11 | 3.76E-12 | 2.65E-11 | 1.36E-10 | 4.96E-11 | 4.65E-11 | 4.25E-11 | 4.69E-11 | 3.34E-11 |
| HT_nc | CTUh | 6.43E-11 | 1.79E-11 | 1.09E-10 | 1.86E-10 | 1.55E-10 | 1.46E-10 | 1.34E-10 | 1.49E-10 | 1.07E-10 |
| IR | kBq U-235 eq. | 1.98E-03 | 1.57E-04 | 1.86E-03 | 1.46E-03 | 5.61E-02 | 5.27E-02 | 4.82E-02 | 5.31E-02 | 3.92E-02 |
| LU | Pt | 11.61 | 1.09 | 19.04 | 16.05 | 2.52 | 2.36 | 2.17 | 2.39 | 1.75 |
| ODP | kg CFC11 eq. | 4.21E-09 | 2.95E-10 | 4.69E-09 | 4.01E-09 | 6.16E-08 | 5.78E-08 | 5.29E-08 | 5.83E-08 | 4.26E-08 |
| POF | kg NMVOC eq. | 2.35E-04 | 1.47E-05 | 3.03E-04 | 3.06E-04 | 9.41E-04 | 8.84E-04 | 8.14E-04 | 8.92E-04 | 6.53E-04 |
| FRD | MJ | 0.43 | 0.03 | 0.40 | 0.28 | 6.19 | 5.81 | 5.32 | 5.86 | 4.30 |
| MRD | kg Sb eq. | 4.41E-07 | 4.39E-08 | 5.23E-07 | 2.84E-07 | 3.88E-06 | 3.64E-06 | 3.33E-06 | 3.67E-06 | 2.72E-06 |
| WU | $m^3$ depriv. | 0.16 | 0.02 | 0.26 | 0.01 | 0.35 | 0.33 | 0.30 | 0.33 | 0.25 |

(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

*6.5.6 Comparison of climate change results with previous studies*

The CC impacts assessed in this study were notably higher than those reported in previous studies, reflecting the significant variability in LCA outcomes based on specific production systems. Salomone et al.[22] reported a CC impact of 2.1 kg $CO_2$ eq./kg protein for dried BSFL. Similarly, Komakech et al.[58] found a lower CC score of 1.24 kg $CO_2$ eq./kg BSFL protein meal. Pahmeyer et al.[63] reported 2.77 kg $CO_2$ eq./kg dried BSFL and 6.96 kg $CO_2$ eq./kg BSFL protein, while Smetana et al.[32] presented a wider range of 1.36 – 15.1 kg $CO_2$ eq./kg insect protein meal. In terms of lipids, Salomone et al.[22] also reported a CC impact of 2.9 kg $CO_2$ eq./kg BSFL lipid, which is lower than the findings of this study (2.5 – 17 kg $CO_2$ eq./kg BSFL lipid on dry matter basis).

These substantial differences in results can be attributed to several factors, including differences in system boundaries, methodological choices, assumptions made, and technology and scale of production in the various studies. However, a major contributing factor to the higher CC impact in our study was the reliance on electricity for heating, which was less efficient than alternative energy sources such as residual heat or renewable energy. The use of electric heating systems, particularly for an energy-intensive insect rearing system, contributed significantly to the environmental burden. The variation in CC impact findings across studies emphasizes the importance of standardized methodologies and transparent reporting in LCA studies. These studies demonstrate that BSFL production can achieve a lower environmental impact under optimized conditions and compete with conventional products.

*6.5.7 Sensitivity analysis*

The results showed that electrical energy use was of great concern as it contributed to all the impacts assessed. Electricity was needed for heating during insect breeding and processing activities of milling and sieving. Switching to other energy sources is likely to change impact results. To test this hypothesis, several scenarios regarding the electricity used in the foreground system were tested by performing a sensitivity analysis considering the following changes:

a) Scenario 1 – 10% improvement of the total energy efficiency in the BSFL production system
b) Scenario 2 – alternative source of energy (photovoltaic system).
c) Scenario 3 – an alternative source of energy (natural gas for climate control and electricity from the national grid for milling and sieving).

d) Scenario 4 – an alternative source of energy (biogas for climate control and electricity from the national grid for milling and sieving).
e) Scenario 5 – utilization of residual heat from biogas for climate control and electricity from the national grid for milling and grinding.

A local sensitivity analysis was performed to assess the impact of varying critical parameters on different impact categories. This involved decreasing the input values for electricity consumption (fans, heat pump, mill, and sieve) by 10% at a time and calculating the sensitivity ratio (SR) using the equation:

$$SR = \frac{\frac{\Delta result}{initial_result}}{\frac{\Delta parameter}{initial_parameter}}$$

6.5.7.1 Scenario analysis

From the results obtained, all scenarios demonstrated a net reduction in environmental impacts relative to the base case across all categories, with a few exceptions: MRD in Scenario 2, EFT in Scenarios 3 and 4, and WU in Scenario 4. Scenario 5 achieved the greatest impact reductions among the scenarios, while Scenario 1 showed the least reductions (Table 6.11).

In Scenario 1, a 10% improvement in the total energy efficiency of the BSFL production system resulted in a proportional 10% reduction across all impact categories. This highlights the significant influence of electricity consumption on the overall environmental impact. Implementing strategies to mitigate electricity use is thus critical to enhancing the environmental performance of BSFL production.

Replacing grid electricity with electricity from a three kWp building-integrated photovoltaic system (PV) in Scenario 2 led to substantial reductions in several impact categories, including IR, ODP, WU, LU, FRD, and CC. Notably, CC decreased from 5.63 to 1.27 kg $CO_2$ eq./kg fresh larvae. However, MRD increased by 77% due to material depletion associated with PV panel production. While solar energy significantly improves environmental performance, challenges include the upfront investment cost, seasonal variability (especially in winter), and end-of-life waste management for PV systems.

**Table 6.11.** Scenario analysis - relative change of to the baseline

| Impact | Scenario 1 | Scenario 2 | Scenario 3 | Scenario 4 | Scenario 5 |
|---|---|---|---|---|---|
| AP | -10% | -68% | -74% | -60% | -89% |
| CC | -10% | -77% | -26% | -67% | -88% |
| ETF | -10% | -19% | 42% | 264% | -85% |
| PM | -10% | -36% | -70% | -32% | -84% |
| MEU | -10% | -67% | -71% | -48% | -88% |
| FEU | -10% | -58% | -82% | -61% | -88% |
| TEU | -10% | -69% | -71% | -54% | -89% |
| HT_c | -9% | -8% | -16% | -19% | -83% |
| HT_nc | -10% | -5% | -46% | -23% | -90% |
| IR | -10% | -84% | -82% | -41% | -89% |
| LU | -10% | -81% | -80% | -30% | -86% |
| ODP | -10% | -83% | -45% | 55% | -88% |
| POF | -10% | -63% | -60% | -61% | -88% |
| FRD | -10% | -81% | -27% | -67% | -89% |
| MRD | -10% | 77% | -81% | -41% | -88% |
| WU | -10% | -83% | -89% | 518% | -90% |

(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

Scenario 3 replaced electricity with natural gas for climate control while retaining grid electricity for milling and sieving. This substitution led to a substantial reduction in WU, MRD, IR, and FEU impacts. CC decreased by 26%, from 5.63 to 4.17 kg $CO_2$ eq./kg fresh larvae. However, EFT increased by 42%. Natural gas, commonly used for heating, is generally more cost-effective than electricity via heat pumps, making it a practical alternative. The results suggest that changing the energy source for climate control offers greater impact reductions than a 10% improvement in total energy efficiency.

Using biogas for climate control while relying on grid electricity for milling and sieving (Scenario 4) showed more impact reductions than improving energy efficiency by 10 % (Scenario 1) across most categories. CC reduced from 5.63 to 1.85 kg $CO_2$ eq./kg fresh larvae, surpassing the reductions achieved with natural gas. However, WU increased by 518% and EFT by 264%. These

increases are attributed to the cultivation of energy crops and emissions from the biogas plant. As a renewable substitute for natural gas, biogas represents a sustainable energy option, but its environmental costs, mainly related to water use, should be carefully managed.

Scenario 5, which utilized waste heat from a biogas plant for climate control, achieved the highest reductions in environmental impacts. The heat from biogas plants is often underutilized due to their remote locations, the inefficiency of long-distance heat transport, and the need for significant infrastructure investments. However, BSFL production facilities near biogas plants can effectively use this waste heat, leveraging proximity for easy access to residues and heat. Both activities align with waste management objectives, further enhancing their synergy.

Figures 6.12 and 6.13 compare the CC impacts of the best (Diet 5) and worst (Diet 4) BSFL production scenarios with conventional protein and vegetable biodiesel sources. The results are expressed per kg of protein in feed and per MJ of biodiesel energy, respectively.

Regarding protein comparison (Figure 6.12), for Diet 4, Scenarios 5 and 2 had CC impacts comparable to conventional protein sources. In more detail, Scenario 5 exhibited a lower CC score than soybean meal, but fishmeal and pea protein had an approximately three times lower impact. In the case of Diet 5, Scenarios 5, 2, and 4 were comparable to conventional protein sources. Scenario 5 achieved the lowest CC score among all protein sources, while Scenario 2 showed a lower impact than soybean protein but was about three times higher than fishmeal and pea protein.

Concerning biodiesel (Figure 6.13), soybean had the highest CC score per MJ of fat among the options considered, while palm had the lowest. For Diet 4, Scenario 5, the lowest impact scenario, was comparable to rapeseed and lower than sunflower. However, it still exceeded the impact of palm. Scenario 2, while comparable to sunflower, was significantly higher than rapeseed and palm. For Diet 5, Scenario 5 achieved the lowest CC score among all biodiesel sources, similar to its performance in the protein comparison. Scenarios 2 and 4 were comparable to sunflower and rapeseed but remained higher than palm.

The results underscore the critical role of energy sources in determining the environmental performance of BSFL production. Among the scenarios considered, Scenario 5 demonstrates the highest potential for impact reduction through efficient resource utilization and waste heat recovery. These findings also highlight that the environmental performance of BSFL production can be optimized to achieve competitiveness with conventional protein and biodiesel sources.

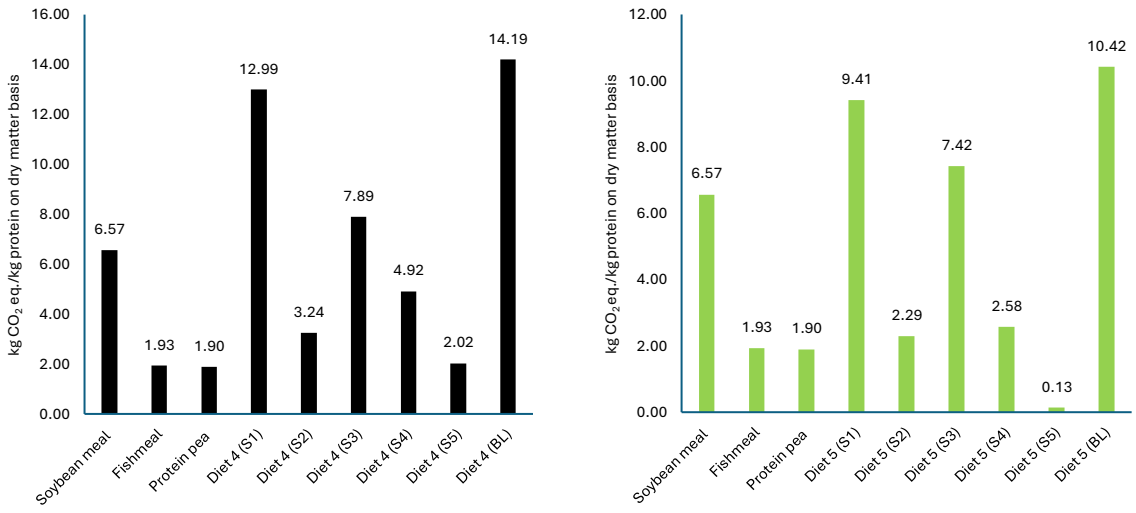



**Figure 6.12**. Climate change impact scenario comparison of worst (Diet 4) and best (Diet 5) BSFL diets vs. conventional protein sources. S – scenario, BL – baseline.

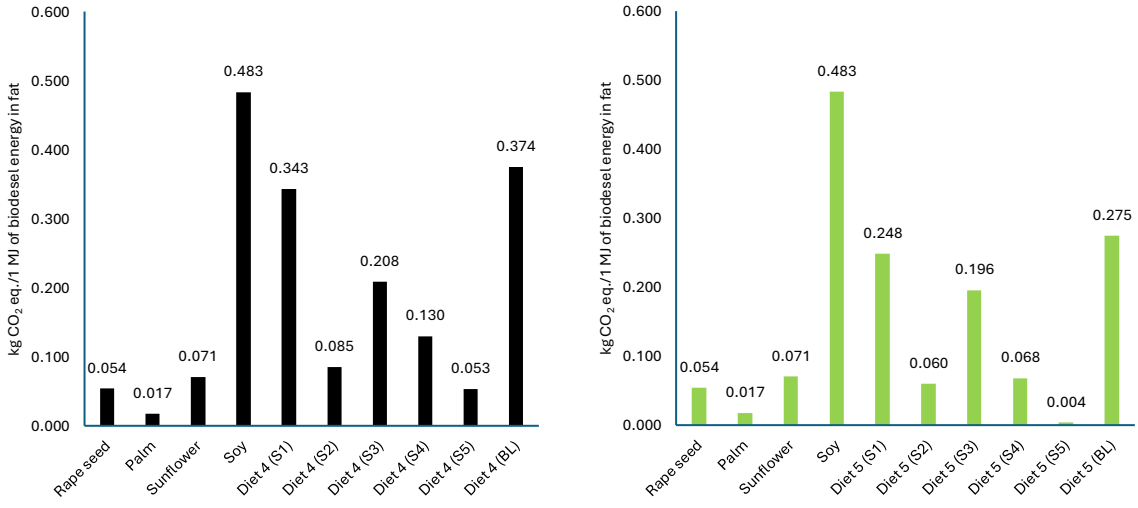



**Figure 6.13**. Climate change impact scenario comparison of worst (Diet 4) and best (Diet 5) BSFL diets vs. conventional vegetable oil for biodiesel production. S – scenario, BL – baseline.

### 6.5.7.2 Perturbation analysis

Given the significant impacts of electricity consumption, a perturbation analysis was conducted to identify the most sensitive model parameters using the sensitivity ratio (SR). The results revealed that electricity consumption by the electric heat pump was the most sensitive parameter (SR > 0.8), followed by electricity use by the mill (SR = 0.12). In contrast, electricity consumption by the fans and sieve had very low or negligible sensitivity ratios (SR < 0.05), as shown in Table 6.12.

As previously discussed, climate control is the primary environmental hotspot in BSFL production due to the high energy demand of the electric heat pump, which is not optimized for the prototype. Since primary data were used in this study, the uncertainty associated with this parameter is low despite its high sensitivity. To mitigate the environmental burdens, implementing strategies to reduce energy consumption, such as replacing the electric heat pump with a low-power air conditioning system, could significantly improve the environmental performance of the BSFL rearing system.

**Table 6.12.** Sensitivity ratio (SR) of selected parameters on electricity consumption.

| Impact | Elec_fan | Elec_heatpump | Elec_mill | Elec_sieve |
|---|---|---|---|---|
| AP | 0.04 | 0.87 | 0.12 | 0.03 |
| CC | 0.04 | 0.86 | 0.12 | 0.03 |
| ETF | 0.04 | 0.83 | 0.11 | 0.02 |
| PM | 0.04 | 0.82 | 0.11 | 0.02 |
| MEU | 0.00 | 0.86 | 0.12 | 0.03 |
| FEU | 0.04 | 0.87 | 0.12 | 0.03 |
| TEU | 0.04 | 0.87 | 0.12 | 0.03 |
| HT_c | 0.04 | 0.81 | 0.11 | 0.02 |
| HT_nc | 0.04 | 0.88 | 0.12 | 0.03 |
| IR | 0.04 | 0.87 | 0.12 | 0.03 |
| LU | 0.04 | 0.84 | 0.12 | 0.02 |
| ODP | 0.04 | 0.86 | 0.12 | 0.03 |
| POF | 0.04 | 0.86 | 0.12 | 0.03 |
| FRD | 0.04 | 0.87 | 0.12 | 0.03 |
| MRD | 0.04 | 0.86 | 0.12 | 0.03 |
| WU | 0.04 | 0.88 | 0.12 | 0.03 |

(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### *6.5.8 Scaling-up considerations (Prospective LCA)*

Given that the data from the mass-scale prototype is based on limited trials and has not yet been fully commercialized, the environmental impacts of BSFL production are expected to change as the system scales. As production expands, changes in efficiency, resource use, and operational dynamics could significantly influence the sustainability of the process.

A key factor in scaling up BSFL production is increasing production yield per tray. Currently, each tray yields approximately 1.5 kg of fresh larvae. However, by optimizing tray utilization, this could potentially increase to 2 – 2.5 kg per tray. At present, only about half of the tray volume is utilized, with 12 kg of substrate per tray, but this could be expanded to 20 kg, improving substrate efficiency and overall productivity. In addition to optimizing individual trays, expanding the number of trays per production unit offers another avenue for scaling. The current mass-scale prototype operates with 132 trays, occupying roughly a quarter of the available container space. By optimizing spatial arrangement, the total number of trays could be increased to 200 – 250 trays, effectively maximizing production capacity within the same footprint.

Beyond physical expansion, automation and process optimization are critical for improving efficiency and sustainability. Implementing automated feeding, monitoring, and harvesting systems could reduce labor intensity, minimize errors, and improve consistency in production. Such technological advancements may lead to lower resource consumption, reduced waste generation, and overall improved environmental performance. From a life cycle perspective, scaling up BSFL production is likely to bring both environmental benefits and trade-offs. Increased production efficiency could lower per-unit impacts related to land use, energy consumption, and emissions. However, higher production volumes may also lead to greater absolute resource use, necessitating careful management of inputs such as fuel for substrate transportation and other consumables. Therefore, as BSFL production scales, continuous environmental monitoring and assessment will be essential to ensure that increased efficiency does not come at the cost of higher overall environmental burdens. Future research should focus on developing optimized production models, integrating renewable energy sources, and exploring closed-loop resource recovery systems to further enhance the sustainability of large-scale BSFL production.

## 6.6 CONCLUSIONS

This study assessed the environmental performance of BSFL production on various diets to valorize agro-industrial residues, focusing on its potential for feed and energy production. An attributional LCA was conducted using primary data from a pilot BSFL rearing system. However, direct gas emissions could not be measured due to sensor challenges.

All tested diets supported BSFL growth, but the diet comprising liquid digestate and wheat residue demonstrated slightly better results in both larvae growth performance and environmental impact than diets consisting solely of agro-industrial residues. The primary environmental hotspot identified was electricity consumption for climate control, contributing over 70% of total impacts across all categories. Regardless of the diet, BSFL production had higher environmental impacts than conventional feed sources such as soybean meal, pea protein, and fishmeal, as well as bioenergy sources like rapeseed, palm fruit, sunflower, and soybean oil. These results underscore the need for significant improvements in heating and energy use for BSFL production to be competitive.

Although current results suggest that BSFL production faces challenges in achieving environmental competitiveness, it remains a promising avenue for sustainable protein for animal feed and fat for biodiesel production. Addressing specific limitations, such as energy use, could enhance its viability. Furthermore, integrating BSFL bioconversion into the horticultural production chain could improve the environmental sustainability of frozen horticultural products. These findings emphasize the potential for BSFL production to contribute meaningfully to circular economy practices when implemented commercially.

The next chapter will culminate the LCA study by evaluating the environmental performance of the entire frozen horticulture production chain, including residue management strategies with BSFL bioconversion and AD, to identify opportunities for enhancing overall sustainability.

# 7 LCA OF FROZEN VEGETABLES WITH RESIDUE VALORISATION

## 7. 1 CHAPTER SUMMARY

This chapter integrates findings from various stages of frozen horticultural production, emphasizing residue management strategies, particularly AD and BSFL biotreatment. The evaluation covered several horticultural products, with detailed cradle-factory-gate analyses conducted for spinach and peas due to data availability. However, only pea residue management with BSFL was considered as at the time of the trials the residues for spinach were unavailable.

The results show spinach production has higher environmental impacts than peas, particularly in CC and processing-related categories, primarily due to higher agrochemical inputs and processing losses. Cultivation was the dominant contributor to most of the environmental impacts for both crops, with energy-intensive processing phases also playing a significant role.

Comparative analyses revealed that AD and BSFL biotreatment reduced the environmental impacts of residue management by substituting conventional products. Scenario 5 of BSFL biotreatment (utilizing residual AD heat for climate control) achieved the lowest environmental impacts, reducing CC impacts by 13% for biodiesel production. Comparing AD and BSFL biotreatment, the environmental performance of AD and BSFL was similar across most impact categories. BSFL offered unique benefits, including the potential use of frass for biogas production and synergy with AD by using liquid digestate as a substrate. Thus, AD's potential for surplus heat utilization and BSFL's ability to generate high-value protein and fat products underscore the complementary nature of these systems.

This chapter highlights the importance of targeted improvements in cultivation, processing, and residue management phases to enhance the sustainability of frozen horticultural supply chains. Integrated approaches leveraging the strengths of both AD and BSFL biotreatment can maximize resource efficiency and environmental benefits.

## 7.2 MODEL FRAMEWORK

This chapter combines the findings from various stages of the frozen horticulture production chain, focusing on residue management strategies involving BSFL bioconversion and AD biotreatment to identify opportunities for enhancing sustainability. The study examined multiple

horticultural products (spinach, peas, tomatoes, and onions) produced under organic and integrated systems. However, due to data limitations, the processing phase was evaluated only for spinach and peas. It is important to note that since integrated production accounted for over 95% of the total output, the processing phase was modeled based on the integrated system.

The processing phase generates a significant amount of residue, with 15% of fresh residue discarded for peas and 30% for spinach. At the BSFL biotreatment phase, spinach residue was not included as no residue was available during the trials. Consequently, this chapter provides a cradle-to-factory-gate analysis for spinach and peas, while residue management using BSFL is only included for frozen peas, compared with AD.

The study evaluates the potential environmental impacts of two residue biotreatment strategies for pea residues in frozen pea production: AD as the baseline and BSFL biotreatment as an alternative. The analysis focuses on two key phases: transportation of the residue to the treatment facility and the treatment process. The system boundaries also include credits for the avoided production of conventional energy sources and materials facilitated by the treatment processes. These avoided products reduce the overall environmental impacts, emphasizing the environmental benefits of residue valorization. However, the system boundaries do not account for the downstream utilization of products generated from the treatment processes.

This comprehensive assessment highlights the potential of residue management strategies to improve the sustainability of frozen horticultural production chains.

## 7.3 BSFL BIOTREATMENT

In the BSFL biotreatment scenario, the system was modeled based on the pea residue + chickpea diet. The primary outputs of this process were larvae and frass. Given their protein and fat content, the larvae were considered a potential replacement for conventional protein sources in animal feed, and the fat was assumed to be a viable alternative for biodiesel production. Based on the physicochemical characteristics of these products (Table 6.4), reference flows were calculated to estimate equivalent substitutions (Table 7.1). Frass was modeled as a 1:1 substitute for chicken manure.

**Table 7.1** Functional units, reference flows, and avoided products in BSFL production.

| Reference substance | Functional unit | Product | Reference flow |
|---|---|---|---|
| Protein | 1 kg | Fresh BSFL | 8.01 kg |
| | | Soybean meal | 2.47 kg |
| | | Pea protein | 5.41 kg |
| | | Fishmeal | 1.74 kg |
| Energy content | 1 MJ/kg fat | Fresh BSFL | 0.206 kg |
| | | Rapeseed | 0.056 kg |
| | | Palm | 0.042 kg |
| | | Soybean | 0.136 kg |
| | | Sunflower | 0.058 kg |

Several scenarios were evaluated for BSFL biotreatment in addition to the baseline, which represents the current production setup:

1. Scenario 1: 10% improvement in energy efficiency.
2. Scenario 2: Use of photovoltaic energy as an alternative to grid electricity.
3. Scenario 3: Use of natural gas for climate control.
4. Scenario 4: Use of biogas for climate control.
5. Scenario 5: Using residual heat from the AD plant for climate control.

These scenarios were designed to explore potential improvements in the environmental performance of BSFL biotreatment and to assess the impact of different energy sources and efficiency measures on the overall system.

## 7.4 ANAEROBIC DIGESTION

As previously stated, the residue from the processing phase is currently sent to an agricultural AD plant. This plant co-digests maize silage, animal manure, and agri-industrial residues to generate biogas for power generation.

The AD plant has a daily capacity of 23.5 MWh of electricity and can process approximately 60 tons of feedstock daily. The biogas produced is used in a combined heat and power (CHP) system to generate electricity and heat. A portion of the electricity is consumed internally for plant operations, while the excess is exported to the grid. The digestate, estimated at 20,000 tons annually, is separated into solid and liquid fractions. Both fractions are stored and applied to fields as organic fertilizer and soil amendment.

An Ecoinvent process was adopted as a baseline to model the AD process for pea residue. Specifically, the process used was "Biowaste {CH}| treatment of biowaste by anaerobic digestion | Cut-off, U," subsequently modified to align with the study's objectives. This proxy process was chosen because it represents typical European AD plants. The system was credited for the electricity generation and digestate production, with substitutions modeled for equivalent electricity from the Italian national grid and chicken manure replaced by the digestate. The amount of substitutable electricity was calculated using the parameters detailed in Table 7.2. For the AD process, 0.18 kg of pea residue was estimated to produce 0.0139 $m^3$ of biogas, with an assumed methane content of 55%. Electricity generation from the biogas was modeled based on a CHP system with an efficiency of 41%, resulting in an estimated 0.033 kWh of electricity substituted. No credit was provided for the heat produced as it is presently not utilized. Regarding the digestate, it was assumed that 90% of the input material by mass was converted to digestate. Transportation of fresh residues was identical to that considered for BSFL biotreatment.

**Table 7.2.** Parameters for pea residue.

| Parameter | Unit | Value |
|---|---|---|
| Water content | % | 79[1] |
| Total solids (TS) | % | 21[1] |
| Volatile solids (VS) | %TS | 96.6[2] |
| Ash | %TS | 3.4[1,3] |
| Methane potential | $m^3$/ton VS | 390[2] |
| Measure methane in biogas | % | 55[4] |

Sources – [1] [252], [2] [253], [3] [254], [4] [21].

## 7.5 RESULTS AND DISCUSSION

### *7.5.1 Cradle-to-factory gate analysis (impacts of frozen horticultural products)*

This LCA study evaluated the environmental impacts of frozen spinach and pea production from cradle to factory gate. The results, summarized in Table 7.3, indicate that pea processing had a lower environmental impact than spinach processing across all impact categories except MEU and EFT. For instance, the CC impact for spinach processing (1017 kg $CO_2$ eq./t product) was approximately double that of pea processing (514 kg $CO_2$ eq./t product). This difference in environmental performance can be attributed to several factors, including higher input requirements and processing losses for spinach. Additionally, the significant difference in processing losses, with spinach experiencing a 30% loss compared to 15% for peas, contributes to the higher environmental impact of spinach processing. However, the higher MEU and EFT scores for frozen peas can be attributed to the higher fertilizer inputs and associated emissions during cultivation.

Strategies aimed at improving both cultivation and processing practices are essential to reduce the environmental burden of frozen vegetable production. Specifically, efforts should be made to optimize crop yields, minimize pesticide and fertilizer use, and reduce waste generation during processing. It is essential to highlight that while spinach and peas are horticultural products, they serve distinct functions and cannot be directly substituted. Therefore, a comprehensive approach that considers each crop's specific characteristics and production systems is necessary to optimize environmental performance.

**Table 7.3.** Cradle-to-factory gate LCIA results for frozen horticultural products (reported per t of frozen product).

| Impact | Unit | Frozen spinach | Frozen pea |
|---|---|---|---|
| AP | mol $H^+$ eq. | 6.59 | 4.09 |
| CC | kg $CO_2$ eq. | 1017.49 | 513.68 |
| ETF | CTUe | 1.27E+05 | 1.42E+05 |
| PM | disease inc. | 3.33E-04 | 2.60E-04 |
| MEU | kg N eq. | 3.68 | 5.55 |
| FEU | kg P eq. | 0.60 | 0.19 |
| TEU | mol N eq. | 1.31E+02 | 1.08E+02 |
| HT_c | CTUh | 5.64E-07 | 4.06E-07 |
| HT_nc | CTUh | 1.65E-05 | 9.69E-06 |

| | | | |
|---|---|---|---|
| IR | kBq U-235 eq. | 4.38E+01 | 16.17 |
| LU | Pt | 3.26E+05 | 1.42E+05 |
| ODP | kg CFC11 eq. | 1.87E-05 | 1.12E-05 |
| POF | kg NMVOC eq. | 4.71 | 3.02 |
| FRD | MJ | 1.32E+04 | 5.02E+03 |
| MRD | kg Sb eq. | 9.86E-03 | 2.93E-03 |
| WU | $m^3$ depriv. | 1146.26 | 704.31 |

(AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### *7.5.2 Hotspot analysis*

#### 7.5.2.1 Cradle-to-factory gate analysis: impacts of frozen spinach

The hotspot analysis for frozen spinach production, as shown in Figure 7.1, revealed that the cultivation phase was the primary contributor to most impact categories, except for WU (18%), IR (20%), ODP (34%), and FRD (37%). At the cultivation stage, agrochemicals (fertilizers and pesticides) and associated emissions, mainly from inorganic nitrogen-based fertilizers, were the main drivers of environmental impact. Mechanized farm activities, such as tillage and irrigation, also contributed to a lesser extent. In the processing phase, energy consumption, primarily from electricity and natural gas, was dominant across several impact categories. The processing phase also made considerable contributions to WU (58%), HT_c (40%), IR (64%), FRD (37%), ODP (34%), and CC (26%). Other production phases had varying levels of influence on the assessed impact categories. Packaging materials accounted for less than 10% of the impacts across all categories except for IR (14%) and FRD (10%), where corrugated boxes were the primary contributors. Transportation of spinach from the farm gate to the factory gate had a minimal impact, contributing less than 10% to all selected midpoint categories.

Comparing the CC results (1017 kg $CO_2$ eq./t frozen product) to previous studies, the results obtained were higher than those reported by Frankowska et al. [200]. They reported lower CC impacts for domestically produced (590 kg $CO_2$ eq./t) and imported (710 kg $CO_2$ eq./t) frozen spinach in the UK, from cradle to factory gate. Their analysis identified the processing phase as the most impactful, followed by cultivation. However, packaging was highlighted as the most

significant impact category for canned spinach, emphasizing the importance of sustainable packaging choices. It is important to note that the results of this study are based on primary data, which can provide more accurate and reliable insights. While, the reliance on secondary data and assumptions in previous studies could introduce uncertainties and limitations in the analysis.

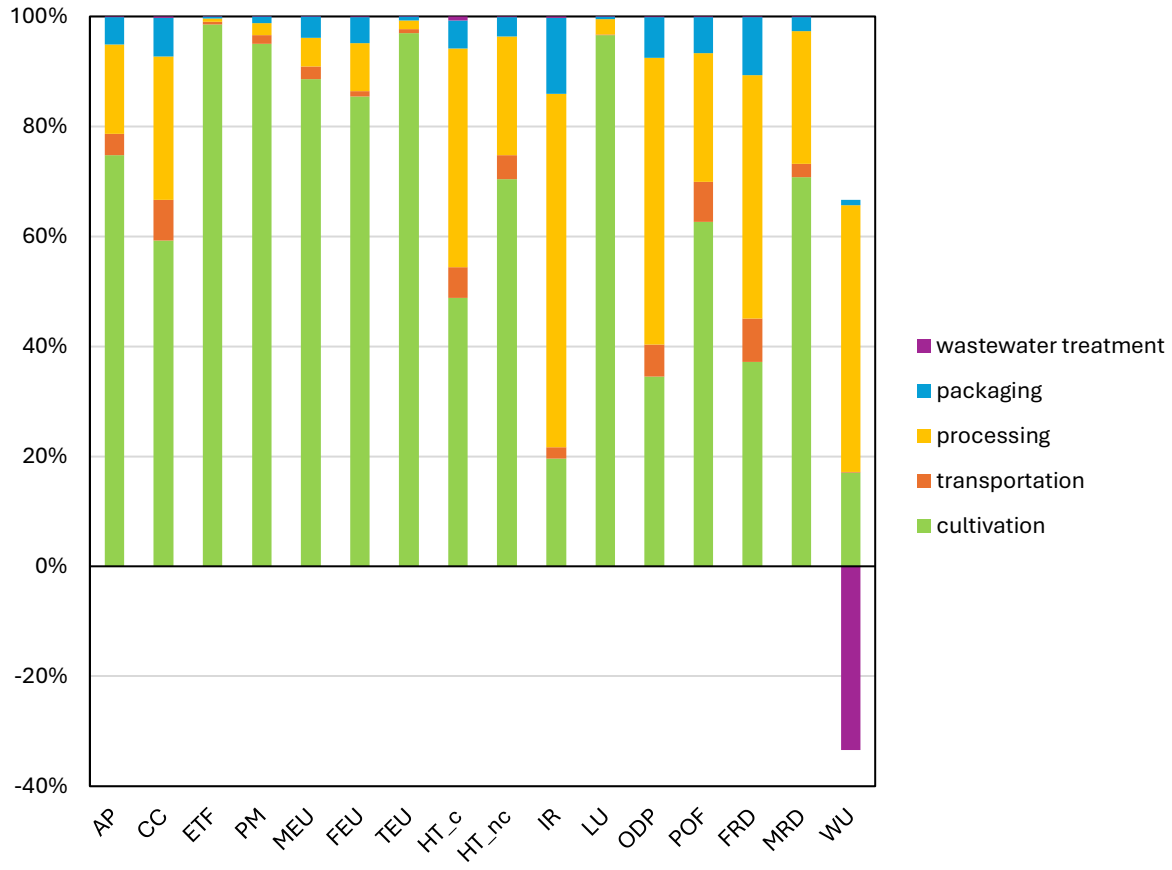



**Figure 7.1.** Hotspots analysis of the phases contributing to the environmental impacts of frozen spinach production. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

### 7.5.2.2 Cradle-to-factory gate analysis: impacts of frozen pea

The hotspot analysis for frozen pea production, illustrated in Figure 7.2, shows that cultivation was the primary contributor to most impact categories, contributing more than 50% toward all impact categories, except WU (9%) and IR (40%). The processing phase also significantly contributed to WU (55%), HT_c (48%), IR (55%), FRD (40%), ODP (30%), and CC (19%). As previously discussed, agrochemical use and associated emissions were the main drivers of impacts at the cultivation stage, while energy consumption

was the primary contributor at the processing stage. Packaging contributed less than 5% to most impact categories, and the impacts from transportation were negligible. This was primarily due to the relatively short transport distances between the fields and the processing facility, with over 60% of the raw materials being sourced regionally and the implementation of efficient packaging practices.

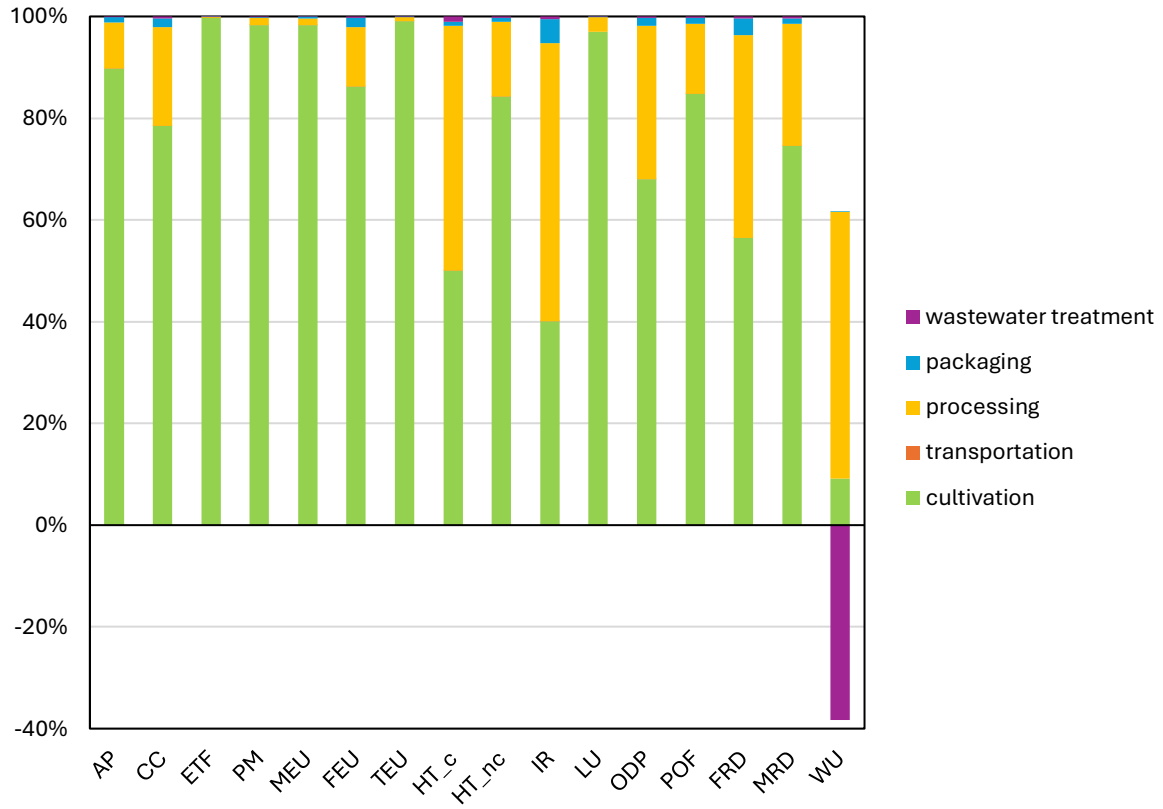



**Figure 7.2.** Hotspots analysis of the phases contributing to the environmental impacts of frozen pea production. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

The calculated CC impact of 531 kg $CO_2$ eq./t frozen peas was lower than the findings from other studies. Del Borghi et al. [201] reported a higher value of 1180 kg $CO_2$ eq./t processed peas, while Frankowska et al. [200] found much higher values for frozen shelled peas produced and imported into the UK (1200 and 1350 kg $CO_2$ eq./t, respectively). The results from this study are also consistent with those reported by Ilari et al. [255] and Ríos-Fuentes et al. [256], which identified cultivation as a key impact category. However, other studies have highlighted packaging and processing as significant contributors to the environmental impacts of frozen vegetable production

[200, 217]. The differences in findings across studies can be attributed to system boundaries, data quality, and regional differences in agricultural practices, processing activities, supply chains, and logistics.

It is important to note that these findings are specific to the particular processing plant and may not represent the entire Italian frozen vegetable supply chain. Transportation quantity, distance, mode, and load capacity can significantly influence the results.

### *7.5.3 Comparison of AD and BSFL bioconversion for residue management in frozen peas production*

A comparative analysis of the various scenarios, including avoided products (Tables 7.4 – 7.10), highlights that the baseline BSFL scenario generally exhibited the highest environmental impact across most categories. In contrast, Scenario 5, utilizing residual heat from the AD plant for climate control, demonstrated the lowest environmental impact. While adding the residue management phase increased the overall environmental impact of frozen peas for some categories, in certain scenarios, BSFL biotreatment led to significant reductions. For instance, Scenario 5 for biodiesel production (with avoided products of soybean and chicken manure) resulted in a 13% reduction in CC impacts associated with frozen pea production. However, no substantial differences were observed between AD and BSFL biotreatment for pea residue management regarding environmental impact.

When evaluating substitution based on protein production, the environmental credits varied depending on the substituted products (Tables 7.4 – 7.6). The soybean meal provided the highest credits for CC and HT_c, while pea protein recorded the highest for MEU, HT_nc, LU, MRD, and WU. On the other hand, fishmeal offered the most credits in AP, FEU, ODP, POF, and FRD. The results showed minimal relative differences when comparing BSFL scenarios to AD, particularly scenario 5, which represents the best-case scenario for BSFL biotreatment. For most impact categories, differences were under 5%, except for MRD, where BSFL scenario 5 was approximately 7% higher. For example, the CC score under soybean meal substitution saw a 3% reduction, while substitutions based on other products showed a slight increase of 1%. AD outperformed BSFL in some impact categories, including FRD, IR, and HT_c. However, BSFL scenario 4 exhibited a higher WU due to the high water involved in biogas production.

In scenarios evaluating biodiesel production, rapeseed provided the highest environmental credits across most impact categories (Tables 7.7 – 7.10). However, soybean performed better in CC and HT_c, while sunflower contributed most to MEU, LU, and MRD. Comparing BSFL scenario 5 to

AD under these options revealed similar results, except in CC for soybean substitution, where BSFL scenario 5 achieved a 14% reduction. As with protein substitution, AD performed slightly better in FRD, IR, and HT_c, while BSFL scenario 4 displayed higher WU.

The relative strengths of AD stem from its capacity to substitute electricity, particularly when the otherwise wasted heat generated is valorized. This underscores AD's potential for even greater environmental savings if the heat is fully utilized. On the other hand, BSFL biotreatment offers unique advantages, particularly in its potential synergy with AD. BSFL had a good growth performance on Diet 5 (liquid digestate + wheat), reducing competition for residue between the two processes. Moreover, frass produced in BSFL treatment, with a methane potential of 315 $m^3$/ton VS, can serve as a viable feedstock for biogas production, further enhancing resource circularity. These findings align with existing studies, which also report minimal differences between AD and BSFL systems, with AD often achieving slightly lower CC impacts [60, 64]. The complementary nature of these two systems highlights the potential for integrated residue management approaches to maximize environmental benefits.

**Table 7.4.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: soybean meal and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 4.06E-03 | 4.67E-03 | 4.23E-03 | 4.19E-03 | 4.11E-03 | 4.26E-03 | 4.04E-03 |
| CC | kg $CO_2$ eq. | 0.52 | 0.62 | 0.61 | 0.52 | 0.57 | 0.54 | 0.50 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.62E-07 | 2.59E-07 | 2.61E-07 | 2.60E-07 | 2.61E-07 | 2.59E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.56E-03 | 5.41E-03 | 5.49E-03 | 5.48E-03 | 5.51E-03 | 5.46E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.36E-04 | 2.31E-04 | 2.12E-04 | 1.99E-04 | 2.11E-04 | 1.98E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.09E-01 | 1.07E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 |
| HT_c | CTUh | 4.06E-10 | 4.15E-10 | 4.12E-10 | 4.16E-10 | 4.04E-10 | 4.14E-10 | 4.01E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.73E-09 | 9.69E-09 | 9.73E-09 | 9.68E-09 | 9.72E-09 | 9.68E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.62E-02 | 3.55E-02 | 1.94E-02 | 1.94E-02 | 2.94E-02 | 1.88E-02 |
| LU | Pt | 1.42E+02 | 1.42E+02 | 1.41E+02 | 1.41E+02 | 1.41E+02 | 1.42E+02 | 1.41E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 3.29E-08 | 3.20E-08 | 1.46E-08 | 2.43E-08 | 4.87E-08 | 1.39E-08 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.32E-03 | 3.27E-03 | 3.11E-03 | 3.09E-03 | 3.13E-03 | 3.03E-03 |
| FRD | MJ | 4.90 | 7.21 | 7.09 | 5.41 | 6.42 | 5.82 | 5.29 |
| MRD | kg Sb eq. | 2.80E-06 | 4.30E-06 | 4.23E-06 | 5.66E-06 | 3.12E-06 | 3.82E-06 | 3.10E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.82 | 0.77 | 0.71 | 0.71 | 1.53 | 0.71 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: soybean meal and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 7.5.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: pea protein and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 4.06E-03 | 4.72E-03 | 4.64E-03 | 4.17E-03 | 4.08E-03 | 4.24E-03 | 4.02E-03 |
| CC | kg $CO_2$ eq. | 0.52 | 0.66 | 0.65 | 0.54 | 0.59 | 0.56 | 0.52 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.62E-07 | 2.62E-07 | 2.61E-07 | 2.60E-07 | 2.61E-07 | 2.59E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.38E-03 | 5.37E-03 | 5.29E-03 | 5.29E-03 | 5.32E-03 | 5.27E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.40E-04 | 2.35E-04 | 2.11E-04 | 1.98E-04 | 2.10E-04 | 1.97E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.09E-01 | 1.09E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.07E-01 |
| HT_c | CTUh | 4.06E-10 | 4.22E-10 | 4.20E-10 | 4.21E-10 | 4.09E-10 | 4.19E-10 | 4.06E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.73E-09 | 9.72E-09 | 9.72E-09 | 9.68E-09 | 9.71E-09 | 9.67E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.82E-02 | 3.60E-02 | 1.92E-02 | 1.92E-02 | 2.91E-02 | 1.85E-02 |
| LU | Pt | 1.42E+02 | 1.40E+02 | 1.40E+02 | 1.39E+02 | 1.39E+02 | 1.40E+02 | 1.39E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 3.48E-08 | 3.24E-08 | 1.42E-08 | 2.38E-08 | 4.83E-08 | 1.35E-08 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.34E-03 | 3.30E-03 | 3.09E-03 | 3.08E-03 | 3.11E-03 | 3.01E-03 |
| FRD | MJ | 4.90 | 7.41 | 7.17 | 5.38 | 6.39 | 5.79 | 5.26 |
| MRD | kg Sb eq. | 2.80E-06 | 4.39E-06 | 4.24E-06 | 5.61E-06 | 3.07E-06 | 3.77E-06 | 3.05E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.76 | 0.75 | 0.65 | 0.64 | 1.46 | 0.64 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: pea protein and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 7.6.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: fishmeal and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 4.06E-03 | 4.67E-03 | 4.59E-03 | 4.11E-03 | 4.03E-03 | 4.19E-03 | 3.97E-03 |
| CC | kg $CO_2$ eq. | 0.52 | 0.66 | 0.65 | 0.54 | 0.59 | 0.56 | 0.52 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.62E-07 | 2.61E-07 | 2.60E-07 | 2.59E-07 | 2.60E-07 | 2.59E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.61E-03 | 5.60E-03 | 5.52E-03 | 5.52E-03 | 5.55E-03 | 5.50E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.37E-04 | 2.32E-04 | 2.08E-04 | 1.95E-04 | 2.08E-04 | 1.95E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.09E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.07E-01 |
| HT_c | CTUh | 4.06E-10 | 4.23E-10 | 4.21E-10 | 4.21E-10 | 4.10E-10 | 4.19E-10 | 4.06E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.74E-09 | 9.74E-09 | 9.74E-09 | 9.69E-09 | 9.73E-09 | 9.69E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.82E-02 | 3.60E-02 | 1.92E-02 | 1.92E-02 | 2.91E-02 | 1.85E-02 |
| LU | Pt | 1.42E+02 | 1.43E+02 | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 2.87E-08 | 2.64E-08 | 8.07E-09 | 1.77E-08 | 4.22E-08 | 7.41E-09 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.32E-03 | 3.28E-03 | 3.08E-03 | 3.06E-03 | 3.09E-03 | 2.99E-03 |
| FRD | MJ | 4.90 | 7.39 | 7.15 | 5.36 | 6.37 | 5.77 | 5.24 |
| MRD | kg Sb eq. | 2.80E-06 | 4.42E-06 | 4.27E-06 | 5.63E-06 | 3.09E-06 | 3.79E-06 | 3.07E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.83 | 0.82 | 0.71 | 0.71 | 1.53 | 0.70 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: fishmeal and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 7.7.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: rapeseed and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 4.06E-03 | 4.56E-03 | 4.48E-03 | 4.01E-03 | 3.93E-03 | 4.08E-03 | 3.42E-03 |
| CC | kg $CO_2$ eq. | 0.52 | 0.66 | 0.64 | 0.54 | 0.59 | 0.56 | 0.51 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.61E-07 | 2.61E-07 | 2.60E-07 | 2.58E-07 | 2.60E-07 | 2.55E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.50E-03 | 5.49E-03 | 5.42E-03 | 5.41E-03 | 5.44E-03 | 5.25E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.40E-04 | 2.36E-04 | 2.12E-04 | 1.99E-04 | 2.11E-04 | 1.93E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.07E-01 | 1.07E-01 | 1.07E-01 | 1.05E-01 |
| HT_c | CTUh | 4.06E-10 | 4.23E-10 | 4.21E-10 | 4.22E-10 | 4.10E-10 | 4.20E-10 | 4.03E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.73E-09 | 9.73E-09 | 9.73E-09 | 9.69E-09 | 9.72E-09 | 9.65E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.83E-02 | 3.61E-02 | 1.93E-02 | 1.92E-02 | 2.92E-02 | 1.79E-02 |
| LU | Pt | 1.42E+02 | 1.41E+02 | 1.41E+02 | 1.40E+02 | 1.40E+02 | 1.41E+02 | 1.40E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 3.51E-08 | 3.28E-08 | 1.45E-08 | 2.41E-08 | 4.86E-08 | 1.29E-08 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.35E-03 | 3.31E-03 | 3.11E-03 | 3.09E-03 | 3.12E-03 | 2.97E-03 |
| FRD | MJ | 4.90 | 7.42 | 7.18 | 5.39 | 6.39 | 5.80 | 5.15 |
| MRD | kg Sb eq. | 2.80E-06 | 4.40E-06 | 4.25E-06 | 5.62E-06 | 3.08E-06 | 3.78E-06 | 2.99E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.81 | 0.79 | 0.69 | 0.69 | 1.51 | 0.64 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: rapeseed and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 7.8.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: palm and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 4.06E-03 | 4.76E-03 | 4.69E-03 | 4.21E-03 | 4.13E-03 | 4.28E-03 | 4.06E-03 |
| CC | kg $CO_2$ eq. | 0.52 | 0.67 | 0.65 | 0.54 | 0.60 | 0.56 | 0.53 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.63E-07 | 2.62E-07 | 2.61E-07 | 2.60E-07 | 2.61E-07 | 2.60E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.62E-03 | 5.61E-03 | 5.54E-03 | 5.53E-03 | 5.56E-03 | 5.51E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.43E-04 | 2.38E-04 | 2.14E-04 | 2.01E-04 | 2.13E-04 | 2.00E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.09E-01 | 1.09E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 |
| HT_c | CTUh | 4.06E-10 | 4.25E-10 | 4.23E-10 | 4.23E-10 | 4.12E-10 | 4.21E-10 | 4.09E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.74E-09 | 9.74E-09 | 9.74E-09 | 9.69E-09 | 9.73E-09 | 9.69E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.86E-02 | 3.64E-02 | 1.96E-02 | 1.96E-02 | 2.95E-02 | 1.89E-02 |
| LU | Pt | 1.42E+02 | 1.43E+02 | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 3.58E-08 | 3.34E-08 | 1.51E-08 | 2.48E-08 | 4.93E-08 | 1.45E-08 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.38E-03 | 3.35E-03 | 3.14E-03 | 3.12E-03 | 3.16E-03 | 3.06E-03 |
| FRD | MJ | 4.90 | 7.48 | 7.24 | 5.45 | 6.46 | 5.86 | 5.33 |
| MRD | kg Sb eq. | 2.80E-06 | 4.47E-06 | 4.32E-06 | 5.68E-06 | 3.14E-06 | 3.84E-06 | 3.12E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.83 | 0.82 | 0.71 | 0.71 | 1.53 | 0.71 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: palm fruit and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 7.9.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: soybean and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol $H^+$ eq. | 4.06E-03 | 4.71E-03 | 4.64E-03 | 4.16E-03 | 4.08E-03 | 4.23E-03 | 4.01E-03 |
| CC | kg $CO_2$ eq. | 0.52 | 0.58 | 0.57 | 0.46 | 0.51 | 0.48 | 0.45 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.62E-07 | 2.62E-07 | 2.61E-07 | 2.59E-07 | 2.61E-07 | 2.59E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.45E-03 | 5.44E-03 | 5.36E-03 | 5.36E-03 | 5.39E-03 | 5.34E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.37E-04 | 2.32E-04 | 2.08E-04 | 1.95E-04 | 2.08E-04 | 1.95E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.09E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.07E-01 |
| HT_c | CTUh | 4.06E-10 | 4.02E-10 | 4.00E-10 | 4.00E-10 | 3.89E-10 | 3.98E-10 | 3.86E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.71E-09 | 9.71E-09 | 9.71E-09 | 9.66E-09 | 9.70E-09 | 9.66E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.83E-02 | 3.62E-02 | 1.93E-02 | 1.93E-02 | 2.93E-02 | 1.87E-02 |
| LU | Pt | 1.42E+02 | 1.40E+02 | 1.40E+02 | 1.40E+02 | 1.40E+02 | 1.40E+02 | 1.40E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 3.51E-08 | 3.28E-08 | 1.45E-08 | 2.41E-08 | 4.86E-08 | 1.38E-08 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.33E-03 | 3.30E-03 | 3.09E-03 | 3.07E-03 | 3.11E-03 | 3.01E-03 |
| FRD | MJ | 4.90 | 7.44 | 7.20 | 5.41 | 6.41 | 5.82 | 5.28 |
| MRD | kg Sb eq. | 2.80E-06 | 4.42E-06 | 4.27E-06 | 5.64E-06 | 3.10E-06 | 3.80E-06 | 3.08E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.83 | 0.82 | 0.71 | 0.71 | 1.53 | 0.71 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: soybean and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

**Table 7.10.** Environmental impacts of frozen pea production: conventional vs. BSFL-based residue valorization (avoided products: sunflower and chicken manure).

| Impact | Unit | FP (AD) | FP (BSFL BL) | FP (BSFL S1) | FP (BSFL S2) | FP (BSFL S3) | FP (BSFL S4) | FP (BSFL S5) |
|---|---|---|---|---|---|---|---|---|
| AP | mol H+ eq. | 4.06E-03 | 4.53E-03 | 4.45E-03 | 3.97E-03 | 3.89E-03 | 4.05E-03 | 3.83E-03 |
| CC | kg CO2 eq. | 0.52 | 0.65 | 0.63 | 0.52 | 0.58 | 0.54 | 0.51 |
| ETF | CTUe | 1.43E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.42E+02 | 1.43E+02 | 1.42E+02 |
| PM | disease inc. | 2.61E-07 | 2.61E-07 | 2.61E-07 | 2.60E-07 | 2.58E-07 | 2.60E-07 | 2.58E-07 |
| MEU | kg N eq. | 5.55E-03 | 5.14E-03 | 5.13E-03 | 5.06E-03 | 5.05E-03 | 5.08E-03 | 5.03E-03 |
| FEU | kg P eq. | 1.93E-04 | 2.38E-04 | 2.33E-04 | 2.09E-04 | 1.96E-04 | 2.08E-04 | 1.95E-04 |
| TEU | mol N eq. | 1.08E-01 | 1.08E-01 | 1.08E-01 | 1.07E-01 | 1.07E-01 | 1.07E-01 | 1.07E-01 |
| HT_c | CTUh | 4.06E-10 | 4.18E-10 | 4.16E-10 | 4.16E-10 | 4.05E-10 | 4.14E-10 | 4.02E-10 |
| HT_nc | CTUh | 9.60E-09 | 9.71E-09 | 9.71E-09 | 9.71E-09 | 9.67E-09 | 9.70E-09 | 9.66E-09 |
| IR | kBq U-235 eq. | 1.48E-02 | 3.80E-02 | 3.58E-02 | 1.90E-02 | 1.90E-02 | 2.90E-02 | 1.84E-02 |
| LU | Pt | 1.42E+02 | 1.38E+02 | 1.38E+02 | 1.37E+02 | 1.37E+02 | 1.37E+02 | 1.37E+02 |
| ODP | kg CFC11 eq. | 1.10E-08 | 3.45E-08 | 3.21E-08 | 1.38E-08 | 2.35E-08 | 4.80E-08 | 1.32E-08 |
| POF | kg NMVOC eq. | 3.01E-03 | 3.30E-03 | 3.26E-03 | 3.06E-03 | 3.04E-03 | 3.08E-03 | 2.97E-03 |
| FRD | MJ | 4.90 | 7.37 | 7.13 | 5.34 | 6.34 | 5.75 | 5.21 |
| MRD | kg Sb eq. | 2.80E-06 | 4.30E-06 | 4.15E-06 | 5.52E-06 | 2.98E-06 | 3.68E-06 | 2.96E-06 |
| WU | $m^3$ depriv. | 0.70 | 0.76 | 0.75 | 0.64 | 0.64 | 1.46 | 0.64 |

FP – frozen pea, AD – anaerobic digestion, BSFL – black soldier fly, BL – baseline, S – scenario. The assessment includes system credits for avoided products: sunflower meal and chicken manure for BSFL-based valorization; electricity from the national grid and chicken manure for AD-based valorization. Data reported per kg frozen peas and based on the system substitution approach. (AP: acidification, CC: climate change, ETF: ecotoxicty freshwater, PM: particulate matter, MEU: eutrophication, marine, FEU: eutrophication, freshwater, TEU: eutrophication, terrestrial, HT: human toxicity (_c: carginogenic, _nc: non-carcinogenic, IR: ionising radiation, LU: land use, ODP: ozone depletion, POF: photochemical ozone formation, FRD: resource depletion, fossils, MRD: resource depletion, minerals and metals, WU: water use).

## 7.6 CONCLUSIONS

This study provides a comprehensive analysis of the environmental impacts associated with frozen horticultural production and explores the potential benefits of alternative residue management strategies, focusing on BSFL biotreatment as an alternative to AD. The findings underscore the importance of optimizing cultivation, processing, and residue management phases to enhance the sustainability of frozen vegetable supply chains.

From the evaluation of the LCA of frozen spinach and peas, the results highlight that spinach production has a significantly higher environmental impact than pea production, primarily due to greater processing losses and higher input requirements during cultivation. Cultivation remained the primary hotspot for most of the impacts assessed.

Both AD and BSFL biotreatment offered some environmental benefits through avoided products such as electricity, soybean meal, chicken manure, and biodiesel feedstocks. Scenario 5 of BSFL biotreatment, which utilized residual heat from the AD plant for climate control, achieved the lowest environmental impact, reducing CC impacts by 13% compared to conventional pea production. Given that the valorization of excess heat from the AD plant was not considered in this study, integrating this heat source into the BSFL rearing system could further enhance the environmental performance of both biotreatment technologies. BSFL demonstrated unique potential for synergy with AD, utilizing liquid digestate as a growth substrate and producing frass with high methane potential. While both systems showed comparable environmental performance, AD slightly outperformed BSFL in certain categories, such as FRD and HT_nc.

Combining AD and BSFL systems can leverage their strengths and mitigate individual limitations, promoting a circular resource and energy recovery approach. Enhancing energy efficiency and valorizing surplus heat can further reduce environmental impacts. Reducing agrochemical inputs, adopting precision farming techniques, and improving crop yields can address the major environmental hotspots identified during cultivation.

This study underscores the potential of innovative residue valorization strategies, such as BSFL biotreatment, to enhance the sustainability of frozen horticultural production. By identifying environmental hotspots and evaluating alternative management options, this research emphasizes the need for holistic and integrated approaches to minimize environmental impacts and maximize resource efficiency.

# 8 GENERAL CONCLUSIONS

This thesis presents a comprehensive analysis of the environmental impacts associated with frozen horticultural production and explores the potential benefits of agro-industrial residue valorization through BSFL bioconversion. Using LCA to evaluate the cultivation, processing, and residue management phases of frozen horticultural supply chains, this research identifies critical environmental hotspots and proposes strategies to enhance sustainability across these systems.

An extensive literature review was conducted to investigate the environmental impacts of BSFL bioconversion. This review included peer-reviewed studies assessing greenhouse gas emissions, ammonia emissions, and other relevant environmental factors, as well as LCA studies examining various aspects of the BSFL value chain. The literature review on BSFL production and its environmental implications revealed the need for real-time gas emission measurements, standardization of methodological approaches, and the establishment of Product Category Rules to assess the environmental impacts of insect-based products. The review identified two primary functions of BSFL systems: waste treatment and biomass production, with energy consumption consistently emerging as a major environmental hotspot.

The cultivation and processing phases analyzed in this study are highly representative of the Italian frozen horticultural production chain, based on data from a leading industry player. Results indicated that integrated farming systems impose a higher environmental burden than organic systems across all categories, mainly due to agrochemical inputs and emissions. However, potential biases arise from the absence of specific processes for organic products and the exclusion of organic farms with zero yields. Processing was another significant contributor, primarily due to energy use for operations and freezing. Additionally, processing generates considerable residues of up to 30% of total output, which, while currently sent to AD, could be further valorized.

To explore the potential of BSFL bioconversion, a prototype rearing system was designed, constructed, and tested to process agro-industrial residues and produce larvae, frass, and eggs. This prototype is one of three currently available on a semi-industrial scale in Italy. Although the system was initially intended to use residual heat from a biogas plant, regulatory and technical challenges led to the installation of an electric heat pump powered by electricity from the biogas plant. While this adjustment was less favorable economically and environmentally, it allowed for semi-industrial-scale testing of BSFL production in Italy. All tested residues, including liquid digestate, demonstrated promising growth performance for BSFL and effective substrate reduction.

However, BSFL production's environmental performance was less competitive than conventional feed sources such as soybean meal, pea protein, and fishmeal, or bioenergy sources like rapeseed and sunflower oil. The high environmental impacts of BSFL production were largely attributed to electricity use for climate control. However, scenarios integrating residual heat from AD into the BSFL system demonstrated significant improvements with comparable impacts to conventional products. Moreover, BSFL also exhibits a synergistic relationship with AD, utilizing liquid digestate as a substrate and producing frass with high methane potential. This integration can enhance resource circularity and reduce environmental impacts by substituting traditional feedstocks like maize silage. To fully realize the potential of BSFL as a sustainable solution, further research is needed to identify optimal production conditions, including energy sources, feedstock types, and geographic location. By addressing energy efficiency and optimizing the production process, BSFL can become a more competitive and sustainable option for producing protein and fat for various applications.

This research emphasizes the need for a holistic and integrated approach to frozen horticultural production, prioritizing interventions like improving nitrogen use efficiency, reducing synthetic fertilizer reliance, and adopting precision farming techniques. For processing, targeted strategies include upgrading equipment, reducing waste generation, and exploring alternative energy sources. Integrating BSFL and AD systems offers a circular resource management approach, generating renewable energy and valuable by-products while minimizing waste.

While BSFL production currently faces challenges in achieving environmental competitiveness, advancements in renewable energy use and optimized production conditions can close this gap. This thesis contributes to sustainable food systems research by highlighting the transformative potential of BSFL and possible AD integration within horticultural supply chains. By addressing the identified challenges, BSFL bioconversion can play a pivotal role in promoting circular bioeconomy practices and reducing the environmental impacts of frozen horticultural production. This work underscores the importance of a systems-based approach to achieve long-term sustainability and meet the growing demand for environmentally sustainable food production.

## 8.1 LIMITATIONS AND FUTURE DIRECTIONS

This study acknowledges several limitations that reflect the emerging nature of BSFL bioconversion technology and the challenges in achieving its full potential. Addressing these

limitations can guide future research and development efforts, ensuring the scalability and sustainability of BSFL production systems.

The BSFL bioconversion system studied remains at the prototype stage, falling short of a fully industrialized system. The system's reliance on manual labor, particularly for feeding, climate control, and harvesting, limits efficiency and increases operational costs. To enable large-scale adoption, a transition to fully automated systems is crucial. Automation can streamline processes, reduce labor intensity, and enhance consistency. Future research should collaborate with engineering and robotics experts to develop cost-effective and scalable automated systems designed specifically for BSFL production.

The study encountered challenges in accurately measuring real-time gas emissions using low-cost sensors due to their limited detection thresholds and interference from the nearby biogas plant. This limitation hindered the precise quantification and monitoring of emissions, which is essential for comprehensive environmental assessments. Investing in high-precision, robust sensors designed for industrial environments is crucial to address this. Collaborative efforts with sensor manufacturers to develop affordable and durable solutions tailored for insect-rearing facilities would significantly improve data reliability.

Another limitation is the inability to fully analyze the biomass characteristics of the diet, larvae, and frass due to analytical constraints. A comprehensive understanding of these characteristics is critical for optimizing feed formulations and enhancing the value of BSFL-derived products. These insights would facilitate the development of tailored diets that improve larval growth and frass quality, thereby boosting the system's overall efficiency and sustainability.

The study did not extend to downstream activities such as testing BSFL-derived products in animal and pet food or extracting functional components like proteins, lipids, and chitin for industrial applications. The environmental impacts and feasibility of such activities on an industrial scale remain unexplored. Future research should include LCAs of these downstream processes to provide a holistic understanding of BSFL's environmental performance. Collaborating with industries that use insect-derived products could enable pilot studies, helping bridge the gap between experimental findings and practical applications.

Economic feasibility remains a critical gap in the study. While environmental assessments are vital, the financial viability of BSFL production is equally essential for its adoption. The current feed cost of approximately EUR 5/kg makes BSFL production non-competitive with conventional

feeds. To address this, studies must evaluate cost-reduction strategies, such as using low-cost feedstocks, improving energy efficiency, and optimizing production processes. Policymakers and industry stakeholders should also consider subsidies or incentives to support the initial adoption of BSFL production systems.

In Italy, industrial-scale BSFL production lags behind other insect-rearing sectors, such as crickets and mealworms. Regulatory challenges hinder development, including prohibitions on using BSFL reared on waste or manure and concerns over safety. Addressing these barriers requires a coordinated effort to establish science-based regulations that ensure safety while enabling innovation. Engaging with regulatory bodies, conducting risk assessments, and developing standardized practices for microbial safety could alleviate these concerns.

Animal welfare issues, while not currently regulated for BSFL, may emerge as a potential obstacle as public awareness grows. Proactively developing welfare guidelines specific to BSFL could preemptively address ethical concerns, ensuring compliance with future regulations.

Recommendations for Future Work

Future studies should focus on overcoming these limitations through the following approaches:

- System scalability: Transition from prototype to industrial-scale systems with automation to improve operational efficiency.
- Advanced low-cost sensor technology: Invest in precise, reliable, real-time gas emission monitoring sensors.
- Downstream LCA studies: Extend environmental assessments to include downstream activities and product applications.
- Economic evaluations: Analyze cost structures, explore subsidies, and identify strategies to lower production costs.
- Regulatory engagement: Advocate for science-based policies to address microbial contamination and welfare concerns, ensuring regulatory alignment for BSFL production.

By addressing these limitations and implementing targeted solutions, the potential of BSFL as a sustainable solution for waste valorization and alternative protein and fat production can be fully realized.

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