SUSTAINABILITY ASSESSMENT OF AGRI-FOOD PROCESSES TOWARDS A CLIMATE NEUTRAL ECONOMY

A Thesis Submitted in Partial Fulfilment of the Requirements
for the Degree of Doctor of Philosophy in

Sustainable Development and Climate change

Doctoral Programme of National Interest

In the Curriculum
AGRICULTURE AND FORESTRY

Stefano Spotorno

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IUSS
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Stefano Spotorno

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ABSTRACT

For a successful transition towards climate-neutral agri-food systems, robust strategies are required to evaluate the mitigation potential and overall environmental impacts of such systems. The European Union, by adopting the Green Deal in 2019, has set the ambitious goal of achieving climate neutrality by 2050. Climate neutrality implies the balance of Greenhouse Gas (GHG) emissions and carbon removals to reach net zero. The agricultural sector is relevant in this area owing to its dual characteristics: carbon sink and source of GHG emissions. Increasing carbon removals from the atmosphere, especially through soil carbon sequestration, is considered as important as reducing GHG emissions from agricultural management operations across the agri-food sector. In recent years, Carbon Farming (CF) has emerged as a promising approach in European policies to increase carbon sequestration in agriculture. The most recent is the Carbon Removal Certification Framework (CRCF), which addresses co-benefits on sustainability and biodiversity in addition to carbon sequestration. In the European context also, circular economy gained attention and needs robust quantitative evaluation. The overarching goal of the thesis is to evaluate the sustainability of agri-food processes through integrated environmental and circularity assessments. To achieve it, the thesis is articulated into five interrelated studies which are used for the methodological integration of soil carbon modelling, LCA, and circular economy assessment in agri-food systems. First, a systematic review highlights the potential and standardization challenges of carbon sequestration in staple crops. Second, Soil Carbon simulations in Flanders demonstrate that while sustainable practices reduce soil carbon emissions, model outputs rely heavily on site-specific conditions and initial data accuracy. Third, coupling LCA with soil carbon modelling in Northern Italy reveals significant trade-offs; CF practices increase carbon sequestration but fertilizers switching does not automatically lower emissions. Notably, organic fertilization maximizes sequestration but increases nutrient leaching compared to reduced tillage. Fourth, a combined environmental and social LCA of circular fertilizers (from sewage sludge ashes) identifies modest environmental gains but substantial social benefits regarding worker safety with respect to traditional fertilizers production. Finally, applying circularity metrics to olive pomace valorisation illustrates how process optimization influences the balance between energy demand and circularity. These results, collectively, bring evidence with respect to the potential and limits of CF, the ecological and social implications of resource recovery, and the worth of the alignment of environmental and circularity metrics, supporting the creation of climate-neutral agricultural food systems that are resource-efficient and socially balanced, taking into account differences in terms of carbon sequestration, resource use and overall environmental performance.

ABSTRACT

Per una corretta transizione verso sistemi agro-alimentari climaticamente neutri, sono necessarie strategie solide per valutare il potenziale di mitigazione e l'impatto ambientale complessivo di tali sistemi. L'Unione Europea, adottando il Green Deal nel 2019, si è posta l'ambizioso obiettivo di raggiungere la neutralità climatica entro il 2050. La neutralità climatica implica l'equilibrio tra le emissioni di gas serra (GHG) e l’assorbimento del carbonio presente in atmosfera per raggiungere lo zero netto. Il settore agricolo assume rilevanza in questo ambito grazie alla sua duplice caratteristica come potenziale serbatoio di carbonio, nei suoli, e come fonte di emissioni di GHG. L'aumento della rimozione del carbonio dall'atmosfera, in particolare attraverso il sequestro del carbonio nel suolo, è considerato importante quanto la riduzione delle emissioni di GHG derivanti dalle operazioni di gestione agricola in tutto il settore agro-alimentare. Negli ultimi anni, il Carbon Farming (CF) è emerso come un promettente approccio per l’aumento del sequestro del carbonio in agricoltura nelle politiche europee. Fra queste il Carbon Removal Certification Framework (CRCF) è il più recente e affronta i co-benefici in termini di sostenibilità e biodiversità oltre al sequestro del carbonio. Nel contesto europeo anche l'economia circolare è stata soggetta a crescente attenzione e necessita di una solida valutazione quantitativa. L'obiettivo generale della tesi è valutare la sostenibilità dei processi agroalimentari. Per raggiungere questo obiettivo, la tesi è articolata in cinque studi interconnessi che vengono utilizzati per l'integrazione metodologica della modellizzazione del carbonio nel suolo, del LCA e metriche per la valutazione dell'economia circolare nei sistemi agro-alimentari. In primo luogo, una revisione sistematica evidenzia il potenziale e le sfide di standardizzazione del sequestro del carbonio nelle colture di base. In secondo luogo, le simulazioni del carbonio nel suolo nella regione delle Fiandre dimostrano che, sebbene le pratiche sostenibili riducano le emissioni di carbonio dal suolo, gli output del modello dipendono fortemente dalle condizioni specifiche del sito e dall'accuratezza dei dati iniziali. In terzo luogo, l'accoppiamento del LCA con la modellizzazione del carbonio del suolo nel Nord Italia rivela compromessi significativi: le pratiche di CF aumentano il sequestro del carbonio, ma il passaggio a fertilizzanti alternativi non riduce automaticamente le emissioni. In particolare, la fertilizzazione organica massimizza il sequestro, ma aumenta la dispersione dei nutrienti rispetto alla lavorazione ridotta del terreno. In quarto luogo, uno studio LCA ambientale e sociale di fertilizzanti da matrice riciclata (ceneri da fanghi di depurazione) identifica modesti vantaggi ambientali, ma sostanziali benefici sociali per quanto riguarda la sicurezza dei lavoratori rispetto alla produzione di fertilizzanti tradizionali. Infine, l'applicazione di metriche di circolarità alla valorizzazione della sansa di oliva illustra come l'ottimizzazione dei processi influenzi l'equilibrio tra domanda energetica e circolarità. Questi risultati, complessivamente, forniscono prove relative al potenziale e ai limiti del CF, alle implicazioni ecologiche e sociali del recupero delle risorse e al valore dell'allineamento dei parametri ambientali e di circolarità, sostenendo la creazione di sistemi agro-alimentari climaticamente neutri, efficienti dal punto di vista delle risorse e socialmente equilibrati, tenendo conto delle differenze in termini di sequestro del carbonio, utilizzo delle risorse e prestazioni ambientali complessive.

LIST OF PUBLICATIONS

This thesis is intended as an “edited collection of articles”, as per IUSS Pavia Doctoral Programme indications. More in detail, the following papers were produced during the PhD Course and are published or under review (at the moment of the submission of the present thesis) in scientific peer-review journals:

- *Arellano Vazquez, D. A., Gagliano, E., Del Borghi, A., Tacchino, V.,* ***Spotorno, S****., & Gallo, M. (2024). Carbon farming of main staple crops: A systematic review of carbon sequestration potential.* ***https://doi.org/10.3390/su16187907***

- **Spotorno, S**., Gobin, A., Vanongeval, F., Del Borghi, A., & Gallo, M. (2024). Carbon farming practices assessment: modelling spatial changes of soil organic carbon in Flanders, Belgium*;* *https://doi.org/10.1016/j.scitotenv.2024.171267*

- ***Spotorno, S****., Gobin, A., Vazquez, D. A. A., Gagliano, E., Del Borghi, A., & Gallo, M. (2025). From Soil Carbon towards System Sustainability: Integrating SOC Modelling and Life Cycle Assessment to evaluate environmental trade-offs in Carbon Farming; https://doi.org/10.1016/j.farsys.2025.100195*

- Under review: Esposito L., Gagliano E., Tacchino V., **Spotorno S**., Canziani R., El Chami D., Gallo M., Del Borghi A., Turolla A. (2026). *Environmental and social sustainability assessment of a circular process for the valorisation of sewage sludge ash and mining by-products into bio-based fertilisers* (*Journal of Cleaner Production*) – February 2026

- Under review: **Spotorno S.,** D’Agostino G., Casazza A., Perego P., Gallo M., Gagliano E., Del Borghi A. (2026*). Life Cycle Assessment and Circularity evaluation of sustainable olive pomace valorization through innovative High-Pressure High-Temperature Extraction* (*Sustainable Materials and Technologies*) – February 2026

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LIST OF ABBREVIATIONS

- **ADPf** - Abiotic depletion potential for fossil resources
- **ADPm** - Abiotic depletion potential for minerals and metals
- **AFOLU** - Agriculture, Forestry and Other Land Use
- **AP** - Acidification Potential
- **BAU** - Business As Usual
- **BECCS** - Bioenergy with carbon capture and storage
- **CAP** - Common Agricultural Policy
- **CB** - Carbon Balance
- **CC** - Cover Crops
- **CED** - Cumulative Energy Demand
- **CDR** - Carbon Dioxide Removal
- **CE** - Circular Economy
- **CF** - Carbon Farming
- **CCS** - Carbon Capture with Storage
- **CCU** - Carbon Capture with Utilization
- **CH_4** - Methane
- **CO_2** - Carbon Dioxide
- **CRCF** - Carbon Removal Certification Framework
- **CS** - Carbon Sequestration

- **DACCS** - Direct air carbon capture and storage
- **E-LCA** - Environmental Life Cycle Assessment
- **E-LCI** - Environmental-Life Cycle Inventory
- **EEA** - European Environmental Agency
- **EPfw** - Eutrophication Potential freshwater
- **EPma** - Eutrophication Potential marine
- **EPte** - Eutrophication Potential terrestrial
- **ET** - Eco-toxicity freshwater
- **EU** - European Union
- **EuFW** - Freshwater Eutrophication
- **EuM** - Marine Eutrophication
- **EuT** - Terrestrial Eutrophication
- **FAO** - Food and Agriculture Organization
- **FW** - Net use of fresh water
- **FYM** - FarmYard Manure
- **GHG** - Greenhouse Gas
- **GLASOD** - Global Assessment of Soil Degradation
- **GWP** - Global Warming Potential
- **GWPbio** - Global Warming Potential biogenic
- **GWPfos** - Global Warming Potential fossil
- **GWPlul** - Global Warming Potential land use and land use change
- **GWPtot** - Global Warming Potential total

- **HPHTE** - High-Pressure High-Temperature Extraction
- **HTc** - Human toxicity cancer effects
- **HTnc** - Human toxicity non-cancer effects
- **HWD** - Hazardous waste disposed
- **IR** - Ionizing radiation, human health
- **LCA** - Life Cycle Assessment
- **LCI** - Life Cycle Inventory
- **LCIA** - Life Cycle Impact Assessment
- **LG-MgO** - Low-Grade Magnesium Oxide mining by-product
- **LU** - Land Use related impacts
- **LULUCF** - Land Use, Land-Use Change, and Forestry
- **MAE** - Mean Absolute Error
- **MRV** - Measurement, Reporting and Verification
- **N_2O** - Nitrous Oxide
- **NECB** - Net Ecosystem Carbon Balance
- **NF_3** - Nitrogen Trifluoride
- **NHWD** - Non-hazardous waste disposed
- **NRSF** - Non-renewable secondary fuels
- **ODP** - Depletion Potential of the stratospheric ozone layer
- **OP** - Olive pomace
- **P** - Phosphorus
- **PENRE** - Non-renewable primary resources used as an energy carrier

- **PENRM** - Non-renewable primary resources with energy content used as material
- **PENRT** - Total use of non-renewable primary energy resource
- **PERE** - Renewable primary resources used as energy carrier
- **PERM** - Renewable primary resources with energy content used as material
- **PERT** - Total use of renewable primary energy
- **PM** - Particulate matter emissions
- **POCP** - Photochemical ozone creation potential
- **PR** - phosphate rocks
- **RMSE** - Root Mean Squared Error
- **RSF** - Renewable secondary fuels
- **RT** - Reduced Tillage
- **RWD** - Radioactive waste disposed
- **S-LCA** - Social Life Cycle Assessment
- **S-LCI** - Social-Life Cycle Inventory
- **SF_6** - Sulfur Hexafluoride
- **SHDB** - Social Hotspots DataBase
- **SDGs** - Sustainable Development Goals
- **SM** - Secondary materials
- **SOC** - Soil Organic Carbon
- **SOM** - Soil Organic Matter
- **SS** - Sewage Sludge

- **SSA** - Sewage Sludge Ash
- **STC** - Soil Total Carbon
- **TSP** - Triple Superphosphate
- **UNFCCC** - UN Framework Convention on Climate Change
- **WDP** - Water deprivation potential
- **WU** - Water Use
- **WW** - Municipal Wastewater

CHAPTER 1

Literature Review and Research Context

1 LITERATURE REVIEW AND RESEARCH CONTEXT

1.1 INTRODUCTION

Climate change is the most pressing and urgent challenge of the modern era. It represents a threat to global stability as it drives long-lasting alterations in weather patterns from the poles to the tropics (Abbass et al., 2022). Climate change is driven by the anthropogenic enhancement of the greenhouse effect due to rising concentrations of greenhouse gases (GHGs). GHGs cause the greenhouse effect in the atmosphere. Greenhouse gases were named as such due to a mechanism that relies on the phenomenon where high-energy short-wave radiation (ultraviolet rays and visible rays) is freely admitted into the atmosphere. These rays warm the outer surface of the earth and are emitted as low-energy long-wave radiation (infrared rays) in all directions. These rays include the surface of the earth. These gases have the capacity to absorb the outgoing radiation and send them in all directions including the surface of the earth. This enhances the reduction in the emission of heat from the surface in direction to space. These gases are defined as Greenhouse Gases (GHGs) and are: Carbon Dioxide (CO_2), Methane (CH_4), Nitrous Oxide (N_2O), Hydrofluorocarbons (HFCs), Chlorofluorocarbons (CFCs), Perfluorocarbons (PFCs), Sulfur Hexafluoride (SF_6) and Nitrogen Trifluoride (NF_3). The greenhouse effect, per se, as a natural effect, is fundamental to guarantee life on Earth, without it the average planet's temperature would be around - 18 °C instead of 15°C. Nevertheless, human activities have significantly increased the concentration of GHGs and altered climate patterns on the planet (Kweku et al., 2018).

Consequences of climate change are mainly related to stresses induced in different sectors, where it acts as a multiplier of these stresses. In particular, the relationship of climate change with the agricultural sector is uniquely complex. Agriculture is one of the main causes of climate change- acting as one of the major drivers of environmental pressures on Earth system safety boundaries (Campbell et al., 2017) - and it is also highly affected by climate change itself, suffering from shifts in temperature or precipitation and from extreme weather events. More in detail, climate change affects agriculture primarily through water and heat stresses, enhancing biotic pressures and increasing yield variability. Such impacts are highly location and crop-specific, usually negatively affecting farm income and productivity. Temperature rise generally reduces yields because it accelerates phenology, precipitation shifts have an effect on timing and variability because of water availability during growing season and extreme events can drive disproportionate damages to farm productivity (Malhi et al., 2021).

All these impacts are mainly related to food security, but climate change has also an effect on biodiversity. Biodiversity is fundamental in agriculture because it sustains ecosystem services which make productive systems more resilient and stable (Bellard et al., 2012). Biodiversity is declining and habitats being altered causing impacts which are primarily related to the shift of species suitable climates, increasing extinction risk which depends strongly on migration capacity (Muluneh, 2021).

As per climate impacts, also overall environmental impacts of agriculture can be highly site-specific, as well as time dependent, showing a non-negligible spatial variability across locations and seasonal fluctuations. Eutrophication and acidification are often recognized as important environmental impacts of the agricultural sector, mainly from nitrogen emissions, ammonia volatilization and nutrient losses - all related to fertiliser management (Lee et al., 2020).

Climate change is also likely to intensify several negative impacts of agriculture, reinforcing the link between the sector and the climate system. Three main connected mechanisms can contribute to this loop: increased nutrient losses on farm, adaptation to stresses, which requires higher water and energy supplies, and non-guaranteed yields may worsen the impacts per unit of product - requiring land expansion (Li et al., 2025; Seppelt et al., 2022). Climate change and socioeconomic trends (population growth, richer diets) can further expand cropland via yield shocks, higher food prices, and shifting land suitability, increasing habitat loss and fragmentation. Biodiversity declines can then also undermine agriculture by reducing the ecosystem services farming relies on. The mechanisms described above suggest the existence of a feedback loop between climate change and agriculture (Yang et al., 2024).

Because of its dual role, agriculture is both a cause of climate change and a source of environmental impacts. Globally, food systems generate around one-third of total anthropogenic GHG emissions (Crippa et al., 2021), taking into account the production alone (the farm-gate) this sector contributes to around 11-13% of total global emissions (Smith et al., 2025). More in detail, agriculture in Europe is among the main sources of GHG emissions, following the energy, industry and transport sectors (EEA, 2025). It is critical to notice the stagnation of emission reductions in the EU agricultural sector since 2005 (**Figure 1.1**). While the EU energy, industry and transport sectors have achieved deep reductions (-30% to -50% since 1990), agricultural emissions significantly decreased between 1990 and 2005 (largely due to the structural collapse of Soviet-era agriculture in Eastern Europe and early efficiency gains) with only a marginal decline of around 2% over nearly two decades (Mielcarek-Bocheńska and Rzeźnik, 2021).

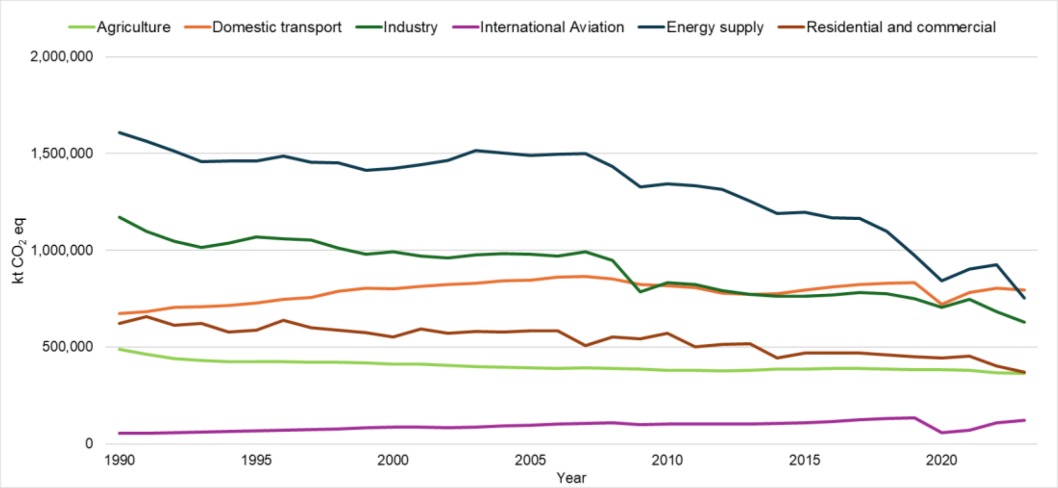

Figure 1-1: *Total greenhouse gas emissions of the EU by sector, based on data reported by EU Member States under the EU Governance Regulation (EEA, 2025).*

In this context, the Agriculture, Forestry and Other Land Use (AFOLU) sector can contribute positively to the mitigation challenge because, despite being a source of GHGs emissions, it has the property of being a potential sink of GHGs through improved land management (Change (IPCC), 2019). This includes a set of options that can contribute to mitigation, such as soil carbon management, but their effectiveness is strongly context-dependent and constrained by issues of measurement and verification, temporal dynamics (including reversibility), and potential trade-offs with other environmental impacts (Schmidt Tagomori et al., 2024).

A comprehensive assessment of sustainability in agri-food systems should ideally integrate environmental, social, and economic dimensions within a unified analytical framework. The economic dimension is particularly relevant in the context of AFOLU contribution to mitigation, where the financial viability of practice adoption is a key determinant of uptake at farm level. The costs associated with transitioning to sustainable management practices must be weighed against potential economic co-benefits such as improved soil fertility, reduced input requirements, and, increasingly, revenues from carbon credit markets under voluntary carbon market (VCM) schemes and emerging regulatory frameworks such as the EU Carbon Removal Certification Framework (CRCF). Preliminary estimates suggest that carbon credit revenues in AFOLU systems may range between €20 and €70 per tonne of CO_2 equivalent sequestered, though transaction costs for measurement, reporting, and verification (MRV) can represent a significant share of total project costs, particularly for smallholder farming systems (Van Hoof, 2023; Piris-Cabezas et al., 2023). Nevertheless, the present thesis deliberately focuses on environmental and social Life Cycle Assessment as its primary methodological framework. Economic assessment - including Life Cycle Costing (LCC) and carbon market feasibility analysis - is explicitly recognised as a complementary analytical dimension and is identified as a priority direction for future research, as discussed in Chapter 8.

1.2 CLIMATE CHANGE MITIGATION AND THE ROLE OF AGRICULTURAL SYSTEMS

As GHG concentrations in the atmosphere and global temperatures continued to increase, a new climate agreement was adopted by the international community in 2015. During the Paris climate conference, the Parties to the UN Framework Convention on Climate Change (UNFCCC) adopted the Paris Agreement. The Paris Agreement is a framework that establishes a global mitigation goal of holding global warming "well below" 2°C while pursuing efforts toward 1.5°C, necessitating both reductions and removals(Delbeke et al., 2019). Parties (i.e. the countries which are part of the Agreement) define mitigation ambitions, implemented through Nationally Determined Contributions (NDCs), and update over time their mitigation measures and targets. In particular, Article 6 of the Paris Agreement is designed to be the legal basis for Parties to set a voluntary international cooperation, in the field of NDCs, allowing for the transfer of mitigation outcomes between countries, thus providing an architecture relevant for the so-called Voluntary Carbon Markets (VCMs) (Piris-Cabezas et al., 2023).

Among the recognized mitigation strategies that generate tradable mitigation outcomes, either under Article 6 cooperative mechanisms or in VCMs, there are avoidance and reduction strategies (known as well as conventional mitigation), removals (carbon dioxide removal or “negative emissions”) and a combination of the two aforementioned strategies to achieve a carbon neutral society (Chen et al., 2022). Conventional mitigation strategies include the use of renewable energy, fuel switching, energy-efficiency improvements, carbon capture with storage (CCS) and carbon capture with utilization (CCU). The mitigation approaches categorized as removals are concerned with the removal of CO_2 from the atmosphere and its storage. The approaches in this category include Bioenergy with carbon capture and storage (BECCS), Biochar, Enhanced weathering, Direct air carbon capture and storage (DACCS), Ocean fertilization/alkalinity enhancement, Soil carbon, Afforestation/Reforestation, Wetland restoration, Mineral carbonation, and the use of biomass in building Some measures deliver a combination of avoided or reduced emissions and removals, depending on system boundaries and accounting (Fawzy et al., 2020).

Under these circumstances, the importance of agricultural systems cannot be overstated, as they have the capability to remove emissions as well as capture emissions through efficient operation and carbon sequestration in the soil and biomass. A broad set of practices can be implemented to reduce emissions and increase removals. Some examples are related to water management change strategies, shifting livestock diets, improving nitrogen-efficiency use, conservation agriculture, precision agriculture and land use efficiency (Pisante et al., 2015). More in detail, production systems become particularly interesting when dealing with sustainable farming practices, to promote soil health and as a pathway to achieve Sustainable Development Goals (SDGs) (Sharma et al., 2024), in this case it is possible to refer to sustainable farming. It is intended as the group of agricultural practices that help not only to maintain but to expand natural resources, protecting the environment and enhancing the efficient use of non-renewable resources. These practices employ methods to produce food, and agricultural products in general, with low environmental impacts avoiding negative effects on the accessibility of food and the use of resources (Chen et al., 2024). The purposes of sustainable farming are to support the agricultural economy, helping to meet world's food needs, improving the environment by ensuring the balance and the correct development of biological cycles (Muhie, 2022).

In this context, the concept of carbon farming emerged as a strategy for the sustainable management of agricultural systems, with the aim of delivering climate mitigation. In scientific literature carbon farming is used as an umbrella term to describe strategies designed to improve the net GHG balance of such agricultural systems from farm to landscape scale (Van Oosterzee et al., 2014).

The practices used to achieve this goal are aimed at the reduction of emissions, the increase of carbon storage in biomass and especially in soils, by augmenting the amount of Soil Organic Carbon (SOC), contributing to CO_2 removal within managed ecosystems (Kumar et al., 2024) and also the reduction of non-CO_2 emissions - as N_2O and CH_4. These effects act through different mechanisms: building soil organic carbon (SOC) by increasing carbon inputs, such as cover crops, improved crop rotations, and residue management; reducing SOC losses through reduced tillage or maintaining soil cover; adding organic carbon via organic amendments, including manure, compost, or digestate; improving nutrient management through precision agriculture and higher fertilizer-use efficiency; optimizing water management through improved irrigation scheduling to increase water-use efficiency and reduce energy demand; integrating trees or other woody elements through agroforestry; improving livestock management through better feed and herd efficiency; and combining strategies, for instance pairing cover crops with reduced tillage.(Maenhout et al., 2024).

Cover crops are non-cash crops grown between main crop cycles (or undersown) to keep the soil biologically active; they add fresh plant material and roots, reduce bare-soil periods, and typically improve soil structure and nutrient retention, which can support higher soil organic carbon over time (Schön et al., 2024). Improved crop rotations increase the diversity and sequencing of crops (often adding legumes, deep-rooted crops, or temporary leys); this tends to diversify root architectures and residue qualities, spread carbon inputs more evenly through time, and can reduce pest and nutrient pressures, lowering reliance on external inputs (De Silva et al., 2026). Residue management refers to how crop residues are handled after harvest (retained, incorporated, mulched, or removed); retaining or mulching residues generally keeps more organic material on-field, protects the surface from erosion, and supplies substrates that can contribute to soil organic matter formation. Reduced tillage (or minimum tillage) decreases the intensity and frequency of soil disturbance; this can help preserve soil aggregates, reduce erosion risk, and limit the rapid breakdown of organic matter that can occur when soil is heavily disturbed, though outcomes depend on climate, soil type, and the full management system. Maintaining soil cover means keeping the soil protected with living plants or crop residues for as much of the year as possible; it reduces erosion and surface sealing, moderates soil temperature and moisture fluctuations, and supports continuous biological activity that helps build and protect soil organic matter (Du et al., 2022). Organic amendments like manure, compost, or digestate add external organic matter and nutrients to soil; they can increase carbon inputs and improve soil structure and fertility, but need careful nutrient management to avoid raising nitrous oxide emissions or nitrate leaching (Biala et al., 2021). Precision agriculture can include both nitrogen and water management, using field data (e.g., sensors, maps, and decision-support) to adjust fertilizer and irrigation rate, timing, and placement to better match crop needs. This can improve nitrogen- and water-use efficiency, reduce energy use for pumping, and help lower nitrogen losses and associated nitrous oxide emissions while sustaining yields (Del Borghi et al., 2022). Agroforestry integrates trees with crops on the same land unit; trees add long-lived biomass carbon, acting as a source of litter and roots that can build soil carbon, and often improve microclimate and erosion control, which can indirectly support both resilience and mitigation (Hernández-Morcillo et al., 2018).

1.3 SOILS AS A CARBON SINK IN AGRICULTURE

As shown in the previous chapter, carbon farming practices are designed to influence soil organic carbon (SOC) levels to provide climate change mitigation. Soils are the largest carbon reservoir on land, and SOC levels reflect a moving equilibrium between carbon inputs and carbon losses. Anthropogenic interventions alter this SOC balance, with consequences for soil fertility and for the atmospheric CO_2 balance-thereby influencing climate change (Lal et al., 2021). In broad terms, carbon farming includes farm management practices on arable land that contribute to mitigation either by enhancing CO_2 removal (sequestering and storing carbon in soils in the form of SOC), or by preventing the loss of carbon already stored in soils (Das et al., 2022) or reducing GHG emissions by influencing biogeochemical processes through land management.

The process of SOC accumulation is due to an excess of carbon input to the soil with respect to carbon losses over multi-year time horizons. Carbon inputs to the soil are crop roots, crop residues and amendments while carbon losses are due to microbial mineralization of carbon, erosion and leaching. SOC persistence is caused by stabilization mechanisms which are in the form of physical protection inside aggregates and the association with other minerals, which can reduce the effect of microbes and enzymes on carbon (Xu and Tsang, 2024). Some of the aforementioned practices, such as cover crops and reduced soil disturbance, can increase SOC pools by increasing carbon inputs and allowing stabilization mechanisms (Fohrafellner et al., 2024).

Therefore, carbon farming acts directly on one of the main biogeochemical cycles: the carbon cycle. The carbon cycle represents the processes through which carbon moves in the major pools - soils, oceans, atmosphere, vegetation and geosphere - and is transformed from organic to inorganic forms and vice versa (Basile-Doelsch et al., 2020). In agricultural systems, CO_2 in the atmosphere is absorbed by plants through photosynthesis, being allocated in plant's biomass as carbon and then it is transferred in soils via litter, crop residues roots and rhizodeposition. The carbon which reached the soil is decomposed and returns partly to the atmosphere as CO_2 via microbial respiration, another fraction is transformed instead and can be stabilized when physically protected or chemically stabilized in aggregates or associated with minerals (Lal et al., 2021).

Carbon farming can hence increase the fraction of incoming carbon which is stabilized and decrease the fraction which is emitted as CO_2 after mineralization. Evidence shows that practices like cover crops can increase SOC in different pools by providing increased carbon inputs and by covering the soil helping stabilization. Another important feature of soil carbon as a sink is permanence (or persistence): it depends on maintaining the same practices over time and on local-context factors which may alter carbon fluxes (Leifeld, 2023).

Among the organic amendments discussed in the context of carbon farming, compost and biochar represent two strategies with particularly high expectations regarding soil carbon sequestration, though through distinct mechanisms and with different degrees of stability. Compost - produced from the aerobic decomposition of organic materials including crop residues, food waste, and manure - provides readily available organic carbon and nutrients to the soil, improving soil structure and microbial activity (Kelbesa, 2021). Its contribution to long-term SOC accumulation is positive and its supply leads to positive effects on C and N accumulation and stabilization also in the deeper layers favoring the increase of long-term soil fertility and C storage (Gioacchini et al., 2024). Biochar, in contrast, is a carbon-rich solid material produced from the pyrolysis of biomass under oxygen-limited conditions. Its highly condensed aromatic structure confers exceptional chemical recalcitrance, resulting in mean residence times in soils that can range from decades to millennia depending on feedstock, pyrolysis temperature, and pedoclimatic conditions (Lehmann et al., 2021). This persistence makes biochar one of the few carbon farming strategies with genuine negative emission potential: global assessments estimate a technical sequestration potential of approximately 0.7 Pg C yr^{-1} from crop residue-derived biochar alone (Karan et al., 2023). Beyond carbon storage, field studies consistently report co-benefits of biochar application including increased soil microbial biomass carbon (+21% on average; Kumar et al., 2025), improved water retention, enhanced cation exchange capacity, and reductions in nitrous oxide (-18%) and methane emissions from soils (Shrestha et al., 2023). However, significant uncertainties remain regarding the net carbon balance of biochar systems, particularly in relation to priming effects on native SOC - which can be either negative (protective) or positive (destabilising) depending on site-specific conditions - as well as production costs, feedstock logistics, and potential ecotoxicological risks (Han et al., 2025). Current process-based SOC models such as RothC lack a standardised parameterisation for biochar inputs, which represents a key methodological challenge for its integration into carbon crediting frameworks and LCA-based assessments. Both compost and biochar are recognised within the EU Carbon Removal Certification Framework (CRCF) as eligible practices for soil carbon removal, but the development of robust, context-specific MRV protocols for these amendments remains an open research priority.

1.4 EUROPEAN UNION POLICY CONTEXT AND THE CRCF FRAMEWORK

The growing consciousness of the climate issue and the recognition of agricultural soil as a critical carbon sink have stimulated the European Union (EU) to develop a comprehensive policy framework to combat climate change and enable soil carbon sequestration as a climate mitigation strategy. This framework is based on a number of interconnected pillars related to climate ambition, food security, soil protection, circular resource management and certification mechanisms.

First, in December 2019 the European Commission adopted the European Green Deal. The main goals of this strategy were to transform Europe into a fair and economically sound society by making the economy competitive while still protecting citizens from environmental harms. Among the aforementioned environmental harms some climate objectives were set. In particular, climate related goals are net climate neutrality by 2050 (translating the Paris agreement's 1.5 °C goal into regional policy), a 55% GHG reductions with respect to 1990 levels and the decoupling of economic growth from resource use. These goals are to be pursued through a package of initiatives and legislations across key sectors such as energy, buildings, mobility, food systems, agriculture, forestry and other land uses and biodiversity (Fetting, 2020). Unlike energy or transport sectors, agriculture can not rely solely on substitution technologies; instead, in the Green Deal it is recognized that agricultural systems must simultaneously reduce emissions and increase carbon removals through improved land management (Montanarella and Panagos, 2021).

Then, in 2020 the Farm to Fork Strategy implemented Green Deal goals by setting quantitative targets in the agricultural sector: 50% reduction in pesticide use, 50% reduction in nutrient losses (through improved N management), and expansion in organic farming area to 25% by 2030. These targets can influence SOC dynamics through reduced external inputs, increased organic farming, and precision agriculture adoption. The strategy acknowledged that achieving these targets requires transparency in environmental performance across food systems (Wesseler, 2022) - positioning Life Cycle Assessment as essential methodology for monitoring progress.

Meanwhile, in 2021 the European Commission adopted the EU soil strategy for 2030. As a large share of EU soils are unsustainably manged and subject to erosion, this strategy aims at addressing comprehensively the land degradation issue. Soils are recognized as critical infrastructure for climate mitigation, biodiversity conservation, and food security. The strategy identifies soil organic carbon as a key performance indicator for soil health and mitigation outcomes (Panagos et al., 2022).

In this context, for climate neutrality purposes the EU Commission established climate-resilient management through the “Communication on Sustainable Carbon Cycles”, in 2021 (European Commission, 2021). The communication includes three key actions: a drastic reduction in the use of fossil carbon, the recycle and reuse of carbon and the increase of carbon removals from the atmosphere either in ecosystems (e.g. carbon farming) or with industrial solutions. In this communication carbon farming is defined as a “business model for healthier ecosystems” which rewards land managers for the implementation of land management practices which result in the increase of carbon sequestration. It defines the practices of carbon farming, its co-benefits and promotes its upscaling. At its core there is the safeguard of biodiversity and the avoidance of ecosystem deterioration (according to the “Do No Significant Harm” principle). In particular, carbon removals must be defined with care and pitfalls such as non-permanence, leakage, and accounting ambiguities can undermine mitigation claims if monitoring and rules are weak (Don et al., 2024). It also highlights how soil carbon credits can be generated, where robust MRV, additionality, and permanence are required for environmental integrity and for avoiding over-crediting or ineffective mitigation outcomes (Paul et al., 2023).

For such purposes, the 2024 Carbon Removal Certification Framework created for the EU an EU-wide uniform certification architecture for carbon removals, including carbon farming, with an explicit aim to enhance transparency and environmental integrity by stipulating uniform norms concerning monitoring, reporting and verification, and by specifying core quality dimensions, decisive for the credibility of soil carbon crediting schemes - additionality and management of non-permanence or reversal risks (Vidal Morant et al., 2025). About agricultural soils, the EU circular economy agenda (2015-2020) is getting increasingly relevant (Calisto Friant et al., 2021), targeting upstream prevention and downstream valorisation of biomass ad waste streams to minimise waste generation, close nutrient and material loops and increase the overall resource efficiency in agri-food value chains (Del Borghi et al., 2020).

Together, these EU policies require developments to translate climate ambition into methodologies and tools for quantifying both emission reductions and carbon removals in agricultural systems. Another key requirement is to ensure environmental integrity and avoid burden shifting, motivating the combined used of life-cycle assessment methods and process-based approaches to evaluate soil organic carbon dynamics under different management scenarios.

Thus, whereas CRCF defines the regulatory and certification structure for carbon farming, scientifically accurate assessment of its environmental benefits needs a robust and science-based methodology. The methodology with such characteristics is life cycle assessment (LCA). LCA of agricultural systems can be measured in terms of accuracy for its climate and environmental performances.

1.5 CIRCULAR ECONOMY PRINCIPLES AND RESOURCE EFFICIENCY IN AGRI-FOOD SYSTEMS

Net-zero in the agri-food sector by mid-century means a radical transformation of the management and consumption of resources in the value chain. The principles stated in the concept of a circular economy, focusing on waste reduction, material recovery, and closing material loops, are a key enabler for this transformation (Geissdoerfer et al., 2017).

The agri-food sector is responsible for the production of a vast amount of residues and by-products within its value chain (Corrado and Sala, 2018). From the point of view of climate change, the dual issue is how to limit the emissions produced in the production stage and, at the same time, reduce the environmental impacts of residue management as well as resources depletion. CE methods address these two issues simultaneously by thinking of agricultural and processing residues not only as waste but also as resources that can be reused within productive loops (Esposito et al., 2020).

Resource efficiency in the agri-food value chain involves two approaches: resource demand reduction and value recovery from residual biomass. Resource demand reduction involves lower water and energy use, and lower synthetic fertilizer use. On the other hand, value recovery involves bio-compounds, soil amendments, and energy production from residual biomass. When integrated with renewable energy technologies, waste valorisation processes could provide value and mitigate carbon emissions for a given output within the sector. The transition toward climate neutrality requires that CE strategies extend beyond incremental waste reduction to fundamental system redesign. Closed-loop models enable nutrient cycling-where processed residues replenish soil fertility or become feedstock for new products-thereby decoupling agricultural productivity from external synthetic inputs. Such resource cycling directly reduces upstream emissions from fertilizer manufacture and transport while improving system resilience (Costa et al., 2022).

The implementation of CE in agri-food requires integration along the full value chain, from farm-scale residue management to processing innovation and product design for material recovery at an industrial scale. Policy frameworks, investments in infrastructure, and technological development need to be aligned to allow a shift from linear "produce-consume-dispose" systems to CE systems where material flows are optimized in line with the environmental impact across their life cycle. In the context of achieving climate-neutral food systems, CE is a key contributor (Chiaraluce et al., 2021). By simultaneously reducing resource extraction, recovering waste value, and enabling renewable-based production cycles, circular approaches aim to provide the resource efficiency essential for industrial-scale food production to operate within planetary boundaries while meeting growing global demand (Batlles-delaFuente et al., 2022).

1.6 LIFE CYCLE ASSESSMENT

Life Cycle Assessment (LCA) is a methodological framework used to estimate the potential environmental impacts associated with a product, process, or service across its entire life cycle. An LCA does not consider only on the processing stage (core processes), but it also includes upstream activities (e.g. raw materials extraction, packaging production, agriculture) and downstream activities (e.g. use phase or waste disposal). Transport required at each stage is also accounted for as part of the life cycle system. In the impact assessment phase, results are translated into multiple environmental indicators through a set of characterisation models, each expressed in its own reference (equivalent) unit. This allows practitioners to identify “hotspots”, meaning the stages, processes, or activities that contribute most substantially to the overall impacts.

LCA is currently ruled by two international standards:

- ISO 14040 (ISO, 2006a): ‘Environmental management - Life cycle assessment - Principles and framework’;
- ISO 14044 (ISO, 2006b): ‘Environmental management - Life cycle assessment - Requirements and guidelines’.

ISO 14040 provides the principles and the general framework of LCA. ISO 14044 provides the detailed requirements and guidelines for conducting a consistent and transparent LCA study. According to ISO 14040, the LCA framework includes (as reported in **Figure 1.2**): goal and scope definition; life cycle inventory (LCI) phase; life cycle impact assessment (LCIA) phase; and life cycle interpretation phase, including other mandatory elements such as reporting and critical review, LCA limitations, the relationships among phases, and the conditions governing the use of value choices and optional elements.

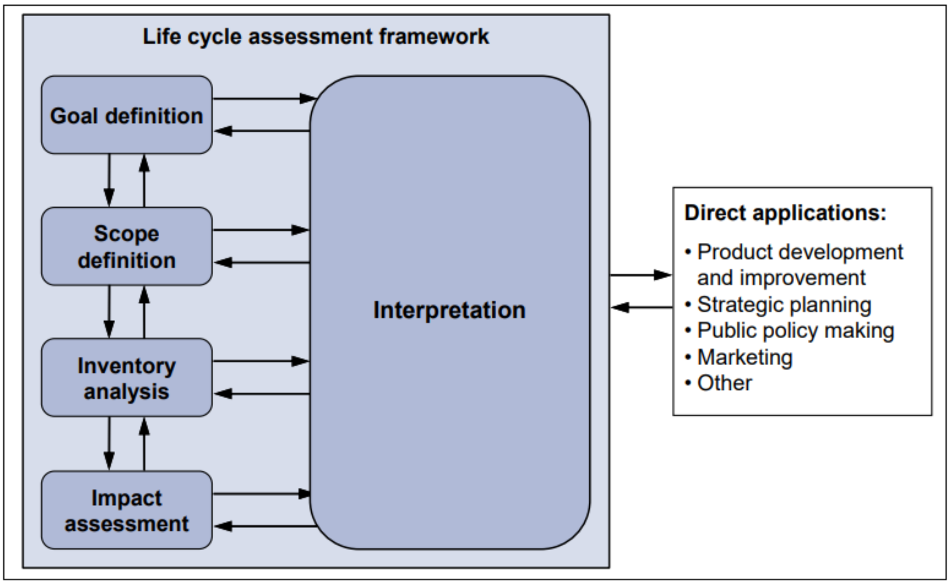

Figure 1-2: Framework for Life Cycle Assessment (JRC, 2010)

In the goal and scope definition phase the purpose of the study is declared. This is often refined iteratively during the analysis, especially when data gaps or methodological constraints emerge. The goal statement should specify the intended application, the reasons for conducting the study, the target audience, and whether results will be used for comparative assertions. The scope then translates everything into a technical plan by describing the product system and its functions, the system boundaries, allocation procedures, chosen impact categories, data quality requirements, and the main assumptions and methodological choices affecting the results.

A fundamental element is the functional unit, which provides the quantitative reference to which all inputs and outputs are normalized, enabling meaningful comparison between systems that deliver the same function. In practice, comparability depends less on the label of the functional unit and more on whether it captures equivalent function and performance, because the functional unit is the anchor that makes results “commensurable” across alternative product systems. For this reason, choices about reference flows, performance characteristics, and any service-life or quality constraints should be aligned with the stated goal, particularly in comparative analysis. System boundaries define which unit processes are included in the model and should cover the full supply chain consistent with the goal, commonly using a cradle-to-grave perspective unless exclusions are justified. Boundaries are often described using the distinction between foreground processes (those directly influenced or modelled in detail) and background processes (those represented through aggregated datasets, such as generic energy or material supply). The product life cycle can also be communicated through upstream, core, and downstream stages-helpful for transparency and hotspot identification-while still ensuring that elementary flows at the interface with nature (resource extraction and emissions to air, water, and soil) are consistently captured as part of the system model.

Multi-functionality requires explicit rules for allocation (partitioning inputs/outputs among co-products), and ISO-based practice prioritizes avoiding allocation where possible before applying allocation relationships. A commonly used hierarchy is: (1) avoid allocation via subdivision of processes or system expansion, (2) allocate using a relevant underlying physical relationship, and (3) if physical causality cannot be established, use another relationship such as economic value. These decisions can materially affect the results, so they must be justified, documented, and kept consistent with the intended application described in the goal (Del Borghi et al., 2014).

The LCI phase compiles and quantifies all relevant input and output flows for the product system, including energy, raw materials, auxiliary materials, and emissions to environmental compartments across the defined system boundary. To connect the model to the functional unit, the inventory must also include product, co-product, and waste quantities so that all flows can be expressed per functional unit and traced back to the reference flows used in the model. Because LCI data may be measured, estimated, or calculated, data quality checks and validation against the stated data quality requirements are essential to ensure fitness for purpose and to support credible interpretation later.

The LCIA phase translates inventory flows into impact category indicators that together form the LCIA profile of the product system, with the selection of impact categories and methods required to be consistent with the goal and scope. Characterisation models (through characterisation factors) convert inventory quantities into common indicator units within each impact category, enabling aggregation of contributions from different elementary flows to a shared impact metric (e.g., within one category). LCIA may be implemented through midpoint or endpoint modelling choices: midpoint methods represent impacts earlier in the cause-effect chain, whereas endpoint approaches model damage at later points (often expressed in terms of human health, ecosystems, and resource availability), and the choice should be transparent and aligned with how results will be used.

The ISO standards also describe optional LCIA elements which may be applied depending on the goal and audience, such as normalisation, grouping, weighting, and data quality analysis. These elements must remain transparent and well documented: when external reference information is used (e.g., for normalisation or weighting), its origin and logic must be reported, allowing users to understand how those choices influence the results and any aggregated indicators.

The interpretation phase integrates findings from LCI and to reach conclusions, explain limitations, and provide recommendations that are consistent with the defined goal and scope. A key requirement is to communicate that LCIA indicators are relative measures of potential impact: they do not predict actual, site-specific damages, threshold exceedances, or risk, but rather support comparative understanding and hotspot identification under the modelling assumptions.

The use of LCA in agricultural systems faces domain-specific challenges, especially with regard to the quantification of dynamic soil carbon pools and their response to management. The section below provides a detailed analysis of SOC dynamics and introduces mechanistic models-through one example, RothC-that allow the LCA practitioner to translate site-level interventions into measurable long-term carbon sequestration outcomes.

1.7 SOIL ORGANIC CARBON (SOC) DYNAMICS AND MODELLING APPROACHES

1.7.1 Overview of Soil Carbon Pools and Turnover Mechanisms

Soil Organic Carbon (SOC) is a key element of the terrestrial carbon cycle and it is the result of a dynamic equilibrium based upon the ratio of carbon inputs (from plant organic material) and carbon outputs (via decompositions and erosions). Soil carbon pools’ structure and function are an essential concept to be understood for predicting the effect of management practices which influence SOC stocks and consequently climate mitigation potential.

SOC can also be conceptually divided, or classified, based on the turnover time which varies according to its stability. The most widely accepted conceptual definition differentiates the three main pools based on the decay rates and the turnover time (decomposition kinetics) (Eriksson, 1971). These pools are:

- The labile pool, which includes fresh plant material (which is readily decomposable) and microbial biomass. Its turnover time is on timescales of years to a few decades and is actively involved in nutrient and carbon cycling.
- The intermediate pool, with intermediate resistance to decomposition (partially stabilized organic matter), is characterised by a turnover time on timescales of decades to a few centuries.
- The stable or resistant pool, which includes highly recalcitrant compounds that are consolidated in aggregates or in minerals, turning over in a timescale which ranges from centuries to thousands of years, conceptualized to function as a long-term carbon reservoir.

Moving on to a more detailed classification, it was conceptualized that the labile fractions of SOC, which includes dissolved organic carbon, readily oxidable carbon, as well as the microbial biomass carbon, can be a sensitive indicator of management practices and of ecosystem disturbances. These fractions are indicators of short-term changes in the carbon cycle and react quickly to agriculture activities such as tillage, residues management, and additions of organic amendments. Mineral-Associated Organic Carbon (MAOC), contributing usually between 80-90% to the mineral soil's total SOC pool, is the most stable fraction and it indicates long-term accumulation of chemically altered organic matter on the surfaces of clays and silt minerals as a result of organo-clay mineral complexation reactions (Kimble et al., 2000).

Two main processes are responsible for the stability of SOC. Firstly, physical stabilization, which entails the physical protection of organic matter in soil aggregates, thus restricting microbial access and consequent decomposition. Secondly, chemical stabilization, which entails the direct interaction of organic matter with the soil minerals, specifically iron, aluminium, and clay, or the formation of complex compounds through the process of condensation, which occurs during the decomposition of the organic matter, thus hindering the microbial conversion of the stabilised carbon to carbon dioxide (Carvalho et al., 2023).

Carbon inputs to the soil are related to complex biological and chemical processes and involve various sources, which include crop and weed residues through incorporation during or at the end of harvest, root exudates and rhizodeposition during plant growth, underground root contribution following plant senescing, and exogenous compost inputs in the form of manure or compost. The extent and nature of carbon inputs and losses drive the processes governing SOC, and practices aimed at either adding carbon or preventing its loss (e.g., conservation tillage and maintenance of soil cover, respectively) modify the balance and provide the opportunity for carbon sequestration in the form of SOC (Kuzyakov and Domanski, 2000).

Due to the complexity and heterogeneity of SOC dynamics, several process-based models have been developed to estimate the dynamics of SOC level changes. The most important difference between SOC models is their model complexity, the number of carbon pools they simulate, the definition of carbon pools, and the way they control the decay rates of carbon through soil and climate and management variables (Paustian et al., 2019).

1.7.2 SOC models and the RothC model

Multiple process-based SOC models exist to predict carbon dynamics in managed systems. Among those most often applied to estimating carbon sequestration in soils are: the Rothamsted Carbon Model (RothC), the DNDC model (short for DeNitrification-DeComposition), DAYCENT, the APSIM model (short for Agricultural Production Systems sIMulator), and the ECOSSE model (short for Estimating Carbon in Organic Soils). While these models vary in terms of complexity, number of carbon pools considered, and level of feedback between carbon and nutrient cycles. DAYCENT extends the CENTURY model framework and includes detailed treatment of plant growth and nutrient limitations, making it suitable for systems where nutrient-carbon interactions are critical. However, all such models require substantial input datasets on soil properties, climate variables, and management practices, which can limit their practical application, particularly when working with climate mitigation purposes or when dealing with regional or spatial scales.

RothC-26.3 is a process-based model specifically designed to simulate organic carbon dynamics in the upper soil horizons (typically 0-30 cm - the so called “topsoil”) of non-waterlogged agricultural soils (Coleman and Jenkinson, 1996). It operates on a monthly time step and represents soil organic matter decomposition through a multi-pool structure that integrates recognized concepts of carbon stabilization and biochemical resistance. In this thesis, the

RothC model was selected for application because it requires a limited number of input parameters that are commonly available from standard soil surveys and remote sensing products, making it well-suited for regional and spatial assessments where data availability is often a constraint (Falloon et al., 2006).

RothC partitions total soil organic carbon into five distinct pools, each with characteristic decomposition rates:

1. Decomposable Plant Material (DPM): Fresh plant residues and easily decomposable organic matter, decomposing with a half-life of approximately 0.1-1 year depending on climate.
2. Resistant Plant Material (RPM): It refers to resistant parts of plants like lignins and cellulose complexes with decomposition half-life of 1 to 5 years.
3. Microbial Biomass (BIO): The living microbial part of the system and the organic matter it contains. Has a turn-over time of 0.5 to 2 years and is an intermediate pool of labile to stable organic matter.
4. Humified Organic Matter (HUM): Chemically altered, decomposing, yet relatively stable organic material, having a half-life of 5 to 50 years.
5. Inert Organic Matter (IOM): Very resistant and non-decomposable organic matter (char and resistant aromatics) that does not decompose and has negligible loss once it is formed.

Allocation of the entering plant material to DPM and RPM fractions is theoretically dependent on the degradability of the entering plant material, usually indicated by the DPM/RPM ratio. A ratio of 1.44 (59% DPM and 41% RPM) is assumed for general crops, whereas crop-specific allocation factors could be determined from laboratory samples.

The rate of decomposition for DPM and RPM is influenced by two major climatic factors, which include temperature and soil humidity. The temperature modifier reflects the exponential increase in microbial metabolic rates with temperature (typically using a Q_{10} approach or similar temperature-response function). The soil water modifier accounts for reduced microbial activity when soils are either too dry (limiting water availability and microbial motility) or waterlogged (creating anaerobic conditions). Additionally, the decomposition rates of all labile and intermediate pools are further modified by soil clay content, which reflects the proportion of carbon physically protected within soil aggregates and mineral-bound organic matter (Pesce et al., 2024). The model can be operated in two modes: the inverse mode, which calculates the annual carbon input required to maintain a given steady-state SOC level (useful for establishing baselines), and the forward mode, which predicts future SOC stocks given defined annual carbon inputs and climate conditions (Nemo et al., 2017). An important aspect of Roth-C is that the inert fraction (IOM) does not change due to management and litter decay, and is determined at the time of initialization, with its equilibrium being that of total SOC and clay content and thereafter does not change. This assumption recognizes that an inert fraction exists that does not affect SOC cycling in the shorter to medium-term time frame, although it does add an element of uncertainty in optimizing the model for soils with non-standard organic chemistry (Gottschalk et al., 2012).

The main inputs required for the model are mean monthly climatic factors (air temperature, precipitation, potential evapotranspiration), soil properties (clay content and initial SOC concentration), and annual inputs of C (amount and DPM/RPM or other measures of degradability). Indeed, the small number of inputs required is one major advantage of the model, as it is relatively easy to approximate these inputs with common soil surveys and climatic measures, as opposed to other process models that simulate more details such as soil water or nutrient competition or microbial populations (Coleman and Jenkinson, 1996).

A relevant methodological consideration in RothC-based forward simulations concerns the treatment of climate inputs. Regional applications of RothC commonly adopt long-term climate averages - typically derived from a recent historical period - held constant across the simulation window, rather than incorporating projected climate trajectories. This assumption of climate stationarity is deliberately adopted to isolate the contribution of land management practices to SOC dynamics, preventing projected climate variability from confounding the attribution of SOC changes to specific carbon farming scenarios. It is furthermore consistent with the requirements of carbon crediting methodologies, which rely on stable, reproducible baseline climate normals to ensure cross-site comparability (Bento et al., 2016; Falloon et al., 2006). However, climate non-stationarity represents a recognised limitation of this approach: rising temperatures and shifting precipitation patterns projected under CMIP6 scenarios are expected to accelerate decomposition rates and alter plant carbon inputs, potentially offsetting part of the sequestration gains estimated in 20-year simulations.

Among the most influential input parameters in RothC simulations is the estimation of carbon inputs to the soil from crop residues, roots, and organic amendments. In regional applications, these values are frequently derived from published experimental studies or default databases rather than from site-specific measurements, introducing a source of structural uncertainty that is difficult to fully propagate through standard sensitivity analyses. In particular, the carbon input increase associated with cover crop adoption is highly variable across pedoclimatic contexts, crop species, and management intensity, with reported values spanning a wide range in the experimental literature (Fohrafellner et al., 2024b; Poeplau and Don, 2015). The use of uniform percentage increases borrowed from studies conducted in different environmental settings represents a recognised limitation of regional RothC modelling and highlights the need for locally calibrated carbon input datasets to improve the robustness and credibility of SOC projections for carbon crediting and MRV purposes.

However, such simplicity also has limitations. RothC does not explicitly simulate the protective effect of reduced tillage, cannot directly model management practices (only their effects on residue carbon inputs and decomposability), and does not account for soil topography or lateral carbon fluxes through erosion and sediment transport. Furthermore, parameter uncertainty-particularly in the estimation of the IOM pool and decomposition rate constants-can propagate through long-term simulations, necessitating validation against independent field observations and long-term experiments. Recent research has demonstrated that RothC performs competitively when validated against long-term field experiments in temperate agricultural systems, with model-predicted SOC changes correlating reasonably well with measured values across diverse soil textures and climate zones. The 20-year simulation timeframe commonly used in carbon crediting schemes aligns well with RothC's simulation window, as the model's greatest predictive confidence lies in medium-term (20-50 year) projections, whereas century-scale projections are more sensitive to assumptions about the stability of the inert pool and the maintenance of specific management practices (Bento et al., 2016).Knowledge gaps and research needs

The above reported literature review allows for the identification of several limitations on the integration of carbon farming, SOC dynamics, LCA and a circular valorisation of agricultural residues.

While the importance of the implementation of carbon farming is becoming widely recognised in regulatory instruments for achieving sectoral climate targets in agriculture (Van Hoof, 2023), there is still an high uncertainty in quantifying SOC changes due to carbon farming practices because of the many context variables and measuring issues, with little alignment between policy needs and models availability to produce robust estimates reproducible on larger extents. To date, limited studies have investigated agricultural management impacts (in terms of SOC accumulation, CO_2 emissions, acidification, eutrophication) with explicit spatial-temporal reference, these effects can instead vary significantly across areas and seasons. This limits the use of results for decision-making at regional scale or for crediting schemes. Also, few studies have combined dynamic process-based SOC (working on multi-annual scales) models with LCA results, usually static.

Scientific literature have the tendency to emphasize de mitigation potential of carbon farming (Cammarata et al., 2025; Fantin et al., 2022) but is not yet comprehensive of trade-off assessment on impact categories which are not climate-related (acidification, eutrophication, resource use) which are relevant for agriculture and strongly related to nitrogen and fertilizers management. Thereby, there is the necessity for a multi-impact assessment to avoid burden-shifting. In the agri-food sector, the transition towards a circular economy requires to fill the gap between “closing loops” and the necessity for environmental performance along the lifecycle, but usually these aspects are treated separately. There is a lack of consolidated approaches to evaluate jointly environmental performances, resource efficiency and design of value chain valorisation in a comparable way.

Thus, in summary, the main research needs identified from the previous discussion are:

- Reduce uncertainty in SOC change quantification through improvements in robustness and reproducibility of model-based estimates at larger spatial scales, which are relevant for MRV, crediting purposes and regional decision-making.
- Develop and apply consistent approaches to integrate dynamic process-based SOC modelling (multi-annual) with LCA frameworks that are often implemented in a more static manner.
- Expand carbon farming assessment beyond climate metrics by adopting multi-impact LCA to capture non-climate trade-offs (e.g., acidification, eutrophication, resource use) strongly influenced by nitrogen and fertiliser management, to avoid burden shifting.
- Bridge circular economy “closing loops” with life cycle environmental performance by establishing consolidated, comparable methods to

jointly evaluate environmental impacts, resource efficiency, and the design of residue-valorisation value chains.

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CHAPTER 2

Design of the Research

2 DESIGN OF THE RESEARCH

2.1 GENERAL RESEARCH FRAMEWORK

Chapter 1 identified critical knowledge gaps requiring a reduced uncertainty in soil organic carbon (SOC) quantification through spatially-explicit modelling, the integration of process-based SOC modelling with Life Cycle Assessment (LCA) frameworks, the expansion of carbon farming assessment beyond climate metrics to capture environmental trade-offs and connection of circular economy principles with lifecycle environmental performance in agricultural systems. This thesis addresses these gaps through a systematic progression from empirical evidence synthesis to regional-scale dynamic modelling, comprehensive multi-impact environmental assessment, and circular economy applications. The research roadmap progresses from understanding carbon sequestration potential in staple crops, to quantifying SOC dynamics through process-based models, to revealing broader environmental effects using an integrated LCA methodology, and finally to include circular economy principles in the agri-food sector both in the upstream process (agriculture: use of circular fertilizers) and in the downstream phase (management of agri-industrial residues to obtain added-value products, minimizing overall environmental burden).

2.2 RESEARCH OBJECTIVES AND THESIS ORGANIZATION

This Ph.D. thesis is presented in a manuscript-based format. The main contents of each chapter are described below:

CHAPTER 1 provides the overview of the study background with particular emphasis on carbon farming, climate change mitigation and circular economy in the agri-food sector.

CHAPTER 2 is about the research objectives, it provides a summarized outline of the thesis, highlighting the original contributions to new knowledge.

CHAPTER 3 is a synthesis of the literature on the rates of carbon sequestration in major staple crops such as maize, wheat, and rice, exploring the gaps in the methods of reporting standardized results. This chapter is based on the publication: *Arellano Vazquez, D. A., Gagliano, E., Del Borghi, A., Tacchino, V.,* ***Spotorno, S.****, & Gallo, M. (2024). Carbon farming of main staple crops: A systematic review of carbon sequestration potential.* ***https://doi.org/10.3390/su16187907***

CHAPTER 4 deals with the evaluation of the spatial variability of SOC changes under carbon farming practices at regional scale using the RothC-26.3 process-based model, assessing model performance and sensitivity, and establish baseline quantification methods suitable for Measurement, Reporting and Verification (MRV) frameworks. This chapter is based on the publication: **Spotorno, S**., Gobin, A., Vanongeval, F., Del Borghi, A., & Gallo, M. (2024). Carbon farming practices assessment: modelling spatial changes of soil organic carbon in Flanders, Belgium*;* ***https://doi.org/10.1016/j.scitotenv.2024.171267***

CHAPTER 5: aims at the integration process-based SOC simulations with multi-impact LCA, assessing not only the sequestration potential of C but also other environmental aspects (acidification, eutrophication, photochemical ozone creation, water use). The aim is to provide information for decision making on the EU's Carbon Removal Certification Framework (CRCF). This chapter is based on the publication: ***Spotorno, S****., Gobin, A., Vazquez, D. A. A., Gagliano, E., Del Borghi, A., & Gallo, M. (2025). From Soil Carbon towards System Sustainability: Integrating SOC Modelling and Life Cycle Assessment to evaluate environmental trade-offs in Carbon Farming;* ***https://doi.org/10.1016/j.farsys.2025.100195***

CHAPTER 6 addresses the environmental and social lifecycle impacts of producing circular phosphorus-based fertilizers through recovery from Sewage Sludge Ash (SSA) and integration with mining by-products, compared to conventional mineral phosphorite-based formulations, and to evaluate whether circular P recovery can maintain or improve environmental and social performance across full supply chains. This chapter is based on the publication under review at the moment of the submission of the present thesis: Esposito L., Gagliano E., Tacchino V., **Spotorno S**., Canziani R., El Chami D., Gallo M., Del Borghi A., Turolla A. (2026). *Environmental and social sustainability assessment of a circular process for the valorisation of sewage sludge ash and mining by-products into bio-based fertilisers* (*Journal of Cleaner Production*).

CHAPTER 7 has the goal to confirm the feasibility of recovering these valuable bioproducts on a large scale by applying innovative High-Pressure High-Temperature Extraction to agro-industrial by-products (olive pomace) and evaluating the circularity of the process as well as material and environmental trade-offs. This chapter is based on the publication under review at the moment of the submission of the present thesis: **Spotorno S.,** D'Agostino G., Casazza A., Perego P., Gallo M., Gagliano E., Del Borghi A. (2026*). Life Cycle*

Assessment and Circularity evaluation of sustainable olive pomace valorization through innovative High-Pressure High-Temperature Extraction (*Sustainable Materials and Technologies*).

CHAPTER 8 presents an overall discussion of the findings with some future perspectives.

CHAPTER 3

Carbon Farming of Main Staple Crops: A Systematic Review of Carbon Sequestration Potential

This chapter is extensively based on the following publication:

Arellano Vazquez, D. A., Gagliano, E., Del Borghi, A., Tacchino, V., ***Spotorno, S****., & Gallo, M. (2024). Carbon farming of main staple crops: A systematic review of carbon sequestration potential.*
https://doi.org/10.3390/su16187907

3 CARBON FARMING OF MAIN STAPLE CROPS: A SYSTEMATIC REVIEW OF CARBON SEQUESTRATION POTENTIAL

Abstract

Carbon farming has become increasingly popular as it integrates agriculture, forestry, and diverse land use practices, all crucial for implementing European strategies aimed at capturing 310 million tons of carbon dioxide from the atmosphere. These farming methods have been shown to increase soil carbon stocks, although results vary by context. However, there is a lack of discussion and consensus regarding the standards used to report these values and their implications. This article analyzes carbon sequestration rates, calculation methodologies, and communication procedures, as well as potential co-benefits and best practices. The average carbon sequestration rates in major staple crops range from very low values (0-0.5 Mg/ha/yr) to medium values (1-5 Mg/ha/yr). Scientific agricultural experiments in key global staple crops demonstrate positive rates of 4.96 Mg C ha-1 yr-1 in wheat-maize rotations and 0.52-0.69 Mg C ha-1 yr-1 in rice-wheat rotations. In agriculture, carbon sequestration rates are reported using different terms that are not consistent and pose communication challenges. This assessment involves a systematic review of the scientific literature, including articles, reviews, book chapters, and conference papers indexed in Scopus from 2001 to 2022. Specifically, this review focuses on long-term experiments, meta-analyses, and reviews that report an increase in soil carbon stock. The research trends observed, through a VOSviewer 1.6.18 analysis, show a steadily increasing interest in the field of carbon sequestration.

3.1 INTRODUCTION

Attaining net-zero CO_2 emissions is essential to keep global warming within 1.5 °C or 2 °C limits (UNFCCC, 2015). In recent years, atmospheric CO_2 levels have reached a record high of 421 parts per million (ppm), representing a 50% increase since the beginning of the industrial revolution (NOAA, 2022). It is estimated that for every 1000 Gt of CO_2 emitted by human activity, the global surface temperature rises by approximately 0.45 °C (Calvin et al., 2023). Mitigation strategies proposed in CO_2 reduction scenarios aim to decrease emissions and enhance carbon dioxide removal (CDR) strategies (Lobus et al., 2023).

In agriculture, CDR strategies aim to increase carbon storage in long-lasting reserves, such as the soil, through the use of plant residues or the accumulation of organic materials (Rodrigues et al., 2023). Soils are one of the two main carbon reservoirs on Earth, holding more carbon than the atmosphere and terrestrial vegetation combined (Rodrigues et al., 2023). Soils represent intricate ecosystems where the equilibrium of soil organic matter (SOM) and living organisms is crucial for decomposing organic substances, recycling minerals, assimilating plant debris, and facilitating plant development (Tian et al., 2022). Nevertheless, SOM loss in agricultural land has resulted in a 50 to 66% reduction in soil fertility compared to past levels, with a decline of 42 to 78 gigatons of carbon, according to historical data (Lal, 2004).

Protecting SOM in agricultural soils is necessary to increase soil carbon stocks and decrease climate change effects in croplands (Grace et al., 2012a). The Global Assessment of Soil Degradation (GLASOD) evaluated thirteen types of degradation, emphasizing wind erosion, water erosion, and physical compaction, as the most impacting in agricultural soils (Carating et al., 2014). These events are worsened by intensive agriculture practices, such as overuse of fertilizers, or the use of intensive tillage (Tziolas et al., 2021). Rattan Lal (Rattan Lal, 2016) highlighted the connection between climate change, agriculture, and soil health and proposed Carbon Farming as a solution to these critical issues, promoting carbon sequestration (CS), crop resiliency, and soil fertility (Sharma et al., 2021).

CS in the Agriculture, Forestry, and Other Land Use (AFOLU) sector can play a significant role in reaching net-zero emissions sooner across multiple global socio-economic trajectories while enhancing existing systems. To achieve this goal, it is crucial to speed up the adoption of carbon sequestration practices in major crops, especially in staple cereals like maize, wheat, and rice-which cover 714 million hectares and represent 32% of the world’s primary crop production, according to FAO (FAO, 2022). Increasing CS in these cereal crops can bring multiple co-benefits, in particular for the Sustainable Development Goals (SDGs) (Bachmann et al., 2022). Lal (Lal, 2004) states that increasing one ton of soil carbon stock in degraded cropland soils may increase crop yield by 20 to 40 kg per hectare (kg/ha) for wheat and 10 to 20 kg/ha for maize.

The carbon content of agricultural lands is usually determined by taking soil samples at a specific depth (e.g., 0-30 cm depth). The samples are then typically analyzed using a dry combustion method, which provides the soil carbon stock for the precise sample location. SOM holds between 55 and 60% of carbon by mass (FAO, 2017). FAO (Millard et al., 2019) proposes a method (Equation 3.1) to find the soil organic carbon (SOC) stock of a sample based on physical measured properties:

$$\mathrm{SOC_i}\ \text{stock}\ (\text{Mg C}\ \mathrm{ha^{-1}}) = (\mathrm{OC_i}) \times (BD\mathrm{fine_i}) \times (1 - \mathrm{vG_i}) \times (\mathrm{t_i}) \times (0.1) \qquad (3.1)$$

where

- $\mathrm{SOC_i}$ = soil organic carbon stock (in Mg C $\mathrm{ha^{-1}}$) of the depth increment i.
- $\mathrm{OC_i}$= organic carbon content (mg C g $\mathrm{soil^{-1}}$) of the fine soil fraction (<2 mm) in the depth increment i.
- $BD\mathrm{fine_i}$ = the mass of the fine earth per volume of fine earth of the depth increment i, (g fine earth [$\mathrm{cm^{-3]}}$) fine earth = dry soil mass [g] - coarse mineral fragment mass [g]) / (soil sample volume [$\mathrm{cm^{-3}}$] - coarse mineral fragment volume [$\mathrm{cm^{-3}}$]).
- $\mathrm{vG_i}$ = the volumetric coarse fragment content of the depth increment i.
- $\mathrm{t_i}$= thickness (depth, in cm) of the depth increment i.
- 0.1 = conversion factor for mg C $\mathrm{cm^{-2}}$ to Mg C $\mathrm{ha^{-1}}$.

In recent years, advancements in elemental analyzers paired with statistical methods brought a decrease to uncertainties associated with SOC sampling, streamlining the process for scientists and farmers by reducing the time and costs of lab work, which is documented across various crop models (Nicoloso et al., 2020).

However, knowing the carbon content of a field is fundamental for assessing CS rates. CS practices include improving crop varieties, improving irrigation strategies, conserving soil moisture, diversifying farming practices, and promoting agroforestry and sustainably sourced agricultural inputs, in contrast with more greenhouse gas-intensive alternatives (Almaraz et al., 2021; Jayaraman et al., 2022). But comprehending and forecasting the potential for carbon sequestration associated with different practices remains a work in progress due to the complexities found within agricultural systems. SOC content shows high spatial and temporal variability. The decomposition process of labile-C and resistant-C converts a fraction of the carbon into the light-C pool, and later, a fraction of the decomposed light-C turns into the heavy-C pool. Eventually, the decomposition of the heavy-C pool produces CO_2, reducing sequestration and increasing by several years the time to process it (Yu et al., 2012).

The objective of this systematic review is to provide readers with an updated overview of CS rates for Carbon Farming practices in maize, wheat, and rice. For this aim, the authors propose a comprehensive analysis of the existing scientific literature and description of the methods, gaps, and trends in practices employed to declare and quantify CS rates. The goal is to encourage further research in the field and promote standardized reporting to ease comparison and data sharing for future experiments, enabling more informed decision-making.

3.2 MATERIALS AND METHODS

In the first step, a systematic review was conducted based on the PRISMA methodology (**Figure 3.1**), using a collection of scientific documents on Carbon Farming studies in the aforementioned cereal staple crops (maize, wheat, and rice) (Page et al., 2021). The documents considered in the analysis (including articles, reviews, book chapters, and conference papers) were published in the Scopus database from 2001 to 2022.

In the document analysis, a four-criteria database search was performed, using the conditions listed below in the queries:

In all queries, the keywords “agriculture” or “carbon sequestration” or “carbon storage” or “C sequestration” or “C storage” were included.

- For the first query, the keywords added were “crops” and “soil”.
- For the second query, the keyword added was “maize”.
- For the third query, the keyword added was “wheat”.
- For the fourth query, the keyword added was “rice”.

A total of 1660 documents were found for the keywords “crops and soil”, 412 for maize, 399 for wheat, and 307 for rice. The selection process is summarized in the PRISMA flow diagram (**Figure 3.1**).

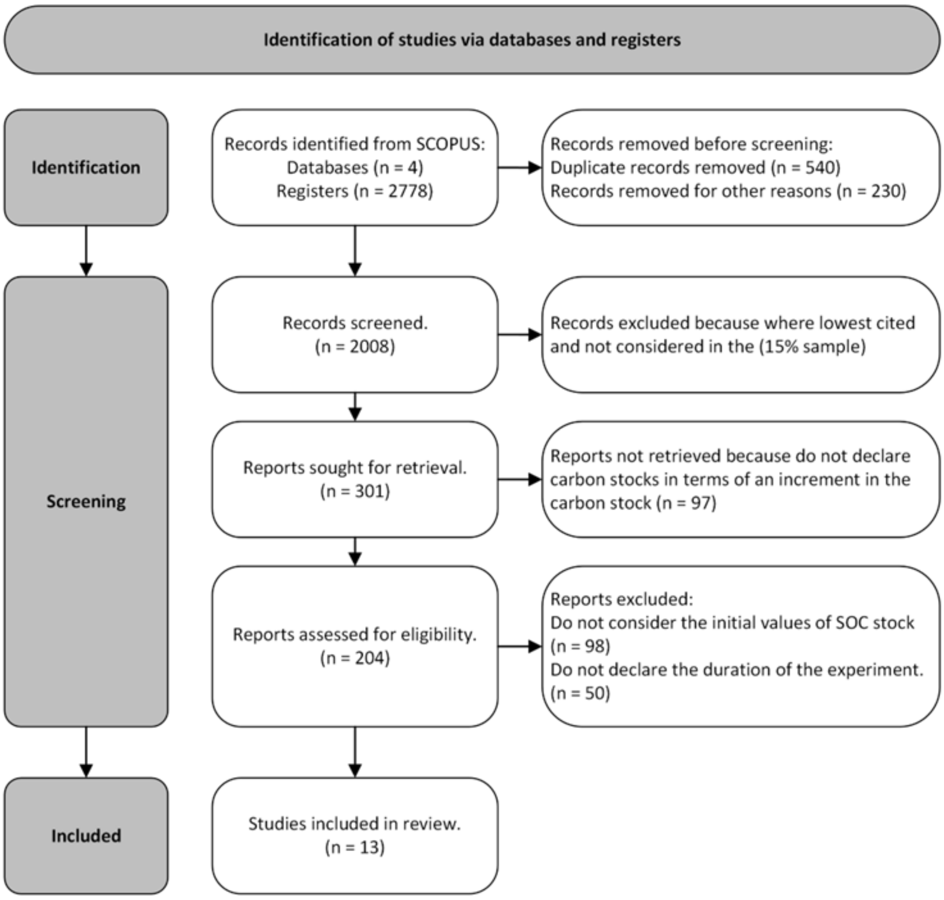

Figure 3-1: Database process of selection based on PRISMA methodology (Page et al., 2021).

Once the documents included were identified, they were divided into two categories:

1. Documents reporting field experiments are commonly based on long-term experiments where CS rates are found through laboratory methods. These methods require soil or biomass samples collected from the field at specific intervals. Variables are shown using analytical machinery or chemical reactions, and samples are taken throughout the experiment (Tirol-Padre et al., 2016).
2. Documents reporting modelling experiments typically involve mathematical-statistical models applied using software simulating physical relationships. These models can simulate various variables,

> such as weather conditions or soil carbon stock flows before the cultivation stage. Data on carbon fluxes and informatics tools are required to conduct these experiments (Kwon et al., 2017).

This review describes the carbon sequestration rates, and the methodologies used for estimation. Additionally, for the selected studies, it provides a brief overview of crop management, the soil profile, the duration of the experiment, and its objectives.

In the next step, to clarify and evaluate the potential impact of these results, the values were classified using the classification proposed by Toensmeier (**Table 3.1**).

Table 3-1: Proposal of ranges to evaluate CS rates in crops by Toensmeier (Toensmeier, 2016)

Classification	Range
Very low	is 0-0.5 Mg/ha/yr.
Low	is 0.5-1 Mg/ha/yr.
Medium	is 1-5 Mg/ha/yr.
High	is 5-10 Mg/ha/yr.
Very high	is 10-20 Mg/ha/yr.
Extremely high	is 20 Mg/ha/yr. or more

At the end, to provide an overview of the practices' trends and co-effects linked to CS practices in the main staple crops listed, a bibliometric analysis was performed using the same databases from Scopus. The co-occurrence bibliometric analysis on all databases listed was conducted using VOSviewer (version 1.6.18). VOSviewer is free software developed by Van Eck and Waltman, which is useful for analyzing bibliometric networks among published papers. VOSviewer was employed to construct bibliometric maps displaying the research trends of CS in the main staple crops (Van Eck and Waltman, 2010).

3.3 RESULTS

3.3.1 Carbon Sequestration rate Analysis

The systematic review of the documents showed that the term "carbon" is used together with other terms and units to report CS rates (**Table 3.2**), which are not consistent. This inconsistency poses a challenge when attempting to compare studies and increases uncertainty about the measurement methods used. For instance, in the study by (Kiran Kumara et al., 2020 the reported carbon values lacked specification in terms of SOC stock. Similarly, in the report by Aller et al., 2018 the increase in SOC stock was not stated in terms of time, making it difficult to understand the temporal aspects of the increase or the relationship between soil, weather, and crop management. To enable a meaningful comparison of documented rates, only the terms that effectively capture changes in carbon stocks over time were selected as functional units for presenting carbon stock changes (**Table 3.2**).

Table 3-2: Definitions of the main terms used in the documents reviewed for CS in the case of its feasibility.

Terms	Units	Definition	Suitable Functional Unit?	References
Annual SOC Sequestered	(Mg C ha^{-1} yr^{-1})	The rate at which soil organic carbon (SOC) is stored in the soil over the long term is typically expressed in terms of years.	Yes	(Liu et al., 2014a)
Average C Input	(Mg C ha^{-1} yr^{-1})	The average carbon content present in biomass residues.	Yes	(Civeira, 2011, p. 40)
Carbon Sequestration Rate (CSR)	(Mg C ha^{-1} yr^{-1})	The rate at which carbon is stored in a reservoir, such as soil or biomass, over a specified period.	Yes	(IPCC, 2019; Saljnikov et al., 2023)
Net Ecosystem Carbon Budget (NECB)	(g C m^{-2} month^{-1})	The difference between the amount of carbon absorbed and released by an ecosystem over a specified period, considering both biotic and abiotic processes.	Yes	(Commission, 2021; Song et al., 2020)
Soil Organic Carbon Sequestration Rates (SOCSRs)	(Mg C ha^{-1} yr^{-1})	The difference between the rates at which organic carbon is stored in the soil over the long term, often due to specific agricultural or forestry practices.	Yes	(Aquino et al., 2017; UNFCCC, 2010)
Soil Total Carbon (STC)	(g C kg^{-1})	The total carbon present in soil, including both organic and inorganic matter.	No	(IPCC, 2019; López-Fando and Pardo, 2011)
Total C Stock (TCs)	(Mg C ha^{-1})	The total carbon stocks in a system, such as an ecosystem or a watershed.	No	(Olorunfemi et al., 2020; UNFCCC, 2010)
Total Carbon	(g C kg^{-1})	The total carbon present in an ecosystem, comprising both biota and soil processes.	No	(IPCC, 2019; Weber et al., 2024)
Total Organic Carbon Pool	(Mg C ha^{-1})	The total amount of organic carbon present in a system, such as an ecosystem or a watershed.	No	(Benbi et al., 2015; IPCC, 2019)

The outcomes presented in **Table 3.2** reveal that, in certain instances, the units employed fail to account for the quantity of carbon sequestered through changes in land management options. For example, the use of the term “carbon pool” may stand for the overall carbon content within a pool, leading to discrepancies. Additionally, variations in the selected time periods result in differences in the measured carbon. Terms such as “equivalent soil mass method”, “soil organic carbon”, “STC”, “TCs”, “total carbon”, or “total organic carbon” contribute to these discrepancies. Lastly, the NECB incorporates emissions from soil mineralization, which falls outside the scope of comparison and data collection parameters in this study.

After considering these results, the following statements were formulated to enhance understanding:

The reviewed documents that do not declare the first C stock, final C stock, and evaluation period were not listed in Appendix A, **Tables A-1** and **A-3**

For an improved comparison, the units used to express CS rates were standardized to Megagrams of carbon per hectare of soil per year (Mg C ha-1 yr-1), with all mathematical conversions performed accordingly (FAO, 2017).

The analysis performed in documents reporting field experiments (Appendix A, **Table A-1**) revealed that CS rates were measured by analytical methods, such as SOC sampling in the top layer of soil, followed by processing the samples with dry combustion in automated analyzers or the Walkley-Black method (a wet chemical oxidation method) between two time periods. Analytical methods were predominant, with only a few instances where biomass was considered as an SOC input. The widely used Walkley-Black method consists in heating SOC and potassium dichromate for the reaction (Walkley and Black, 1934). In the Walkley-Black method, the main disadvantage is incomplete oxidation. In the average oxidation, SOC ranges from 27 to 100% depending on the soil characterization, SOM, and heating method (Shamrikova et al., 2022). In comparison with elemental analyzers, the sum of SOC and soil inorganic carbon could reduce the uncertainty of the measurement (Shamrikova et al., 2023)

Measuring and monitoring stored carbon over time are challenging also due to errors in SOC stock assessment caused by depth measurements over time (Fowler et al., 2023). SOC deposited in topsoil is considered light-C, the youngest and biologically most reactive, with turnover times between a few months and a few years. However, CS requires storing it in long-term secure pools to prevent immediate remission. Nevertheless, light-C must undergo several processes to be transformed into heavy-C forms, which are the most resistant to further degradation and can remain in the soil for hundreds or even thousands of years (Yu et al., 2012). For instance, wind erosion can carry away the light-C layers, while rainfall erosion can wash away biological organic matter, which is necessary to accelerate biomass decomposition (Martín-Lammerding et al., 2013). Additionally, carbon undergoes aerobic processes that release a percentage of the sequestered carbon in the mineralization of soil (Bahri et al., 2019).

The observation period is another parameter in the estimation of CS rates. Most of the analyzed documents were experiments conducted for periods ranging from 3 to 7 years or even over 15 years. The relevance of time was explained in different soil layers by Yu et al., 2012. Changes in land use and/or land management can significantly affect soil carbon stocks. SOC measurements show high spatial and temporal variability in the decomposition process of labile-C, and resistant-C converts a fraction of the carbon into the light-C pool, and later, a fraction of the decomposed light-C turns into the heavy-C pool. Detecting an increase in soil carbon content can take several years, and different approaches are used to quantify carbon removal in current frameworks for Carbon Farming certificates, as acknowledged by the European Commission (Bahri et al., 2019).

In contrast, the analysis performed in documents reporting modelling (Appendix A, **Table A-2**) shows that experiments employ a variety of methods, such as a statistical analysis, Duncan’s Multiple Range Test, and agricultural crop models like CENTURY, Daycent, DSSAT-CERES, SALUS, Roth-C, and APSIM, among others (Nicoloso et al., 2020). These methods rely on statistical techniques that use climate data, soil properties, and crop data based on their phenology from earlier field experiments to simulate longer periods under different management conditions (Spotorno et al., 2024).

The implementation of crop models and laboratory methodologies remains limited due to their intricate nature and challenges in simulating weather conditions and diverse cropping systems, like cover cropping or crop rotation strategies (Le et al., 2018; Sapino et al., 2020). Despite these complexities; long-term experiments have demonstrated the potential of carbon sequestration in staple crops and its advantageous impact (Tong et al., 2014).

In **Table 3.3**, the main differences between field experiments and modelling experiments are listed as a result of the comparison **between Tables A-2 and A-4 of Appendix A**.

Table 3-3: Main differences between analytical and statistical and data modeling methods.

Analytical Methods	Modeling Methods
Mostly based in long-term experiments.	Mostly based in earlier field experiments, and the simulation of long-term periods.
Require land for the experiment and laboratory or machinery equipment.	Require high-performance computing.
Mostly based on laboratory carbon work.	Mostly based on statistical methods and mathematical models.
Mostly require destructive samples of soil.	Require a substantial amount of data of previous field experiments.

Table 3.4 summarizes the average rate of CS of the documents analyzed in Appendix A, **Tables A-1 and A-3**. It is important to note that some studies in Appendix A, **Tables A-1 and A-3**, did not account for soil mineralization, the effects of soil erosion, and the environmental impacts generated by the management. The tables solely declare the potential for sequestration in the measured pool through stock increments.

Table 3-4: Average of CS rates on main staple crops.

Classification	Crop System	CS Potential
Very low	Wheat-Maize (2) * and Maize Monocropping (11) *	0.184-0.454 Mg C ha^{-1} yr^{-1}
Low	Rice-Wheat (6, 7, 8, 13) *	0.52-0.69 Mg C ha^{-1} yr^{-1}
Medium	Wheat-Maize (5, 9) *	4.96 Mg C ha^{-1} yr^{-1}

3.3.2 Research Trends in the Field of Carbon Sequestration through VOSviewer

The database used for the co-occurrence through VOSviewer software was the one resulting from the first query and resulted in 1660 documents. A minimum threshold of 65 was set for the 8526 keywords, and only 100 met the threshold. To cut unrelated words, the repetitive words were merged during a cleaning step before visualizing the map. **Figure 3.2** depicts the obtained map from VOSviewer showing the co-occurrence keywords (depicted as circles), which often appeared in the publications related to CS, while the lines stand for the connection among them. The circle size indicates keyword frequency, and line thickness stands for the strength of connections between keywords, highlighting the number of times they appeared together in the same document.

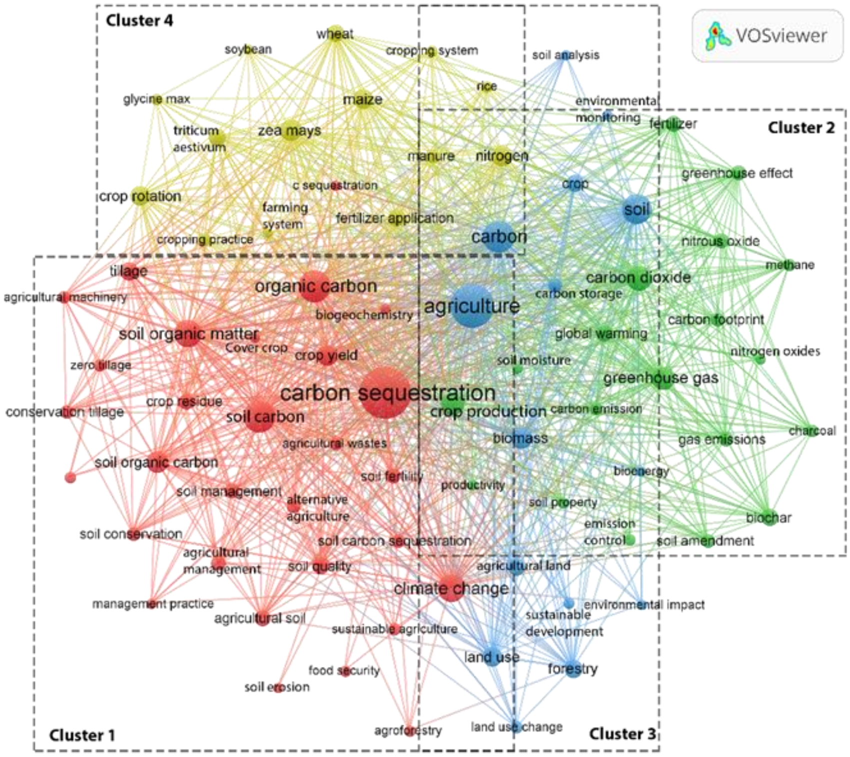

Figure 3-2: Co-occurrence keyword map of CS publications from 2001 to 2022.

Figure 3.2 displays four distinct clusters of keywords. The first cluster, depicted in red, includes keywords related to co-effects of Carbon Farming practices. Keywords such as “soil organic matter“, “soil organic carbon”, “soil fertility”, and “soil erosion” are prominent variables for assessing optimal Carbon Farming practices (Baiamonte et al., 2022). Changes in CS practices are related to keywords such as “agroforestry” and “tillage”, or changes in “agricultural machinery”. These keywords are one of the two remarkable results of the additional analysis performed in Appendix A, **Tables A-2 and A-4.**

The first remarkable result in Appendix A, **Tables A2 and A4**, is tillage reduction, which reduces soil erosion and increases soil microbial activity, which plays a crucial role in converting biomass into SOM (Kirkegaard et al., 2020). The change from plowing to no tillage is remarkable, as it avoids breaking SOM and exposing topsoil SOC to wind or water erosion (Adhikari et al., 2017). This practice, combined with leaving biomass residues on the soil surface after harvesting, proved to be efficient in reducing the need for irrigation, preserving soil moisture, and increasing the amount of organic matter (Tadiello et al., 2022). Consequently, soil with more organic matter can process more carbon effectively (Eddy and Yang, 2022).

In the second cluster, in green, **Figure 3.2** shows the main impacts of intensive agriculture such as air and soil emissions, as indicated by keywords like "greenhouse gases," "methane", "carbon dioxide", "global warming", and "nitrogen oxides" (Zhang et al., 2018).

The third cluster in **Figure 3.2,** represented in blue, holds main carbon pools in ALOFU, "soil", "biomass", and "forestry" and then its relationship with "land use change", and "agriculture", involving the "carbon" and "soil analysis".

The fourth cluster in **Figure 3.2** shows in yellow the key staple crops of this study, "maize (Zea mays)", "wheat (Triticum aestivum)", and "rice", related to the "cropping system", as intercropping o crop rotations, between them or with other crops such as "Soybean (glycine max)". Intercropping o crop rotations included in the cluster conclude another of the two remarkable results of the additional analysis performed in Appendix A, **Tables A-2 and A-4.**

Crop rotation or intercropping are typically planned with a focus on efficiency and economic returns, aiming to increase the yield and reduce the inputs (Woźniak, 2021). Crop rotation combinations, such as maize-wheat and wheat-rice, proved their effectiveness in increasing SOC stock, as described in detail by Clay et al. (Clay et al., 2012). These rotation systems present a promising approach to enhance CS while keeping agricultural productivity (Stern et al., 2012). Intercropping and crop rotation, where two or more crops are grown in the same field area, are typically designed to complement each other in terms of their growth habits and nutrient requirements (Liu et al., 2019). Intercropping helps reduce the external inputs of fertilizer and pesticides and promotes the growth of deep-rooted crops, and, as a consequence, the biomass belowground (Detheridge et al., 2016).

Other words in the cluster such as "manure", "fertilizer application", or "farming system" are also found in the extra analysis performed in Appendix A, **Tables A-2 and A-4.**

Figure 3.3 was created using VOSviewer and highlights the global attention on CS and storage in these main staple crops. The highest link strength is represented in yellow and is based on the same co-occurrence analysis mentioned earlier, considering the occurrences and total link strength of the most frequent keywords.

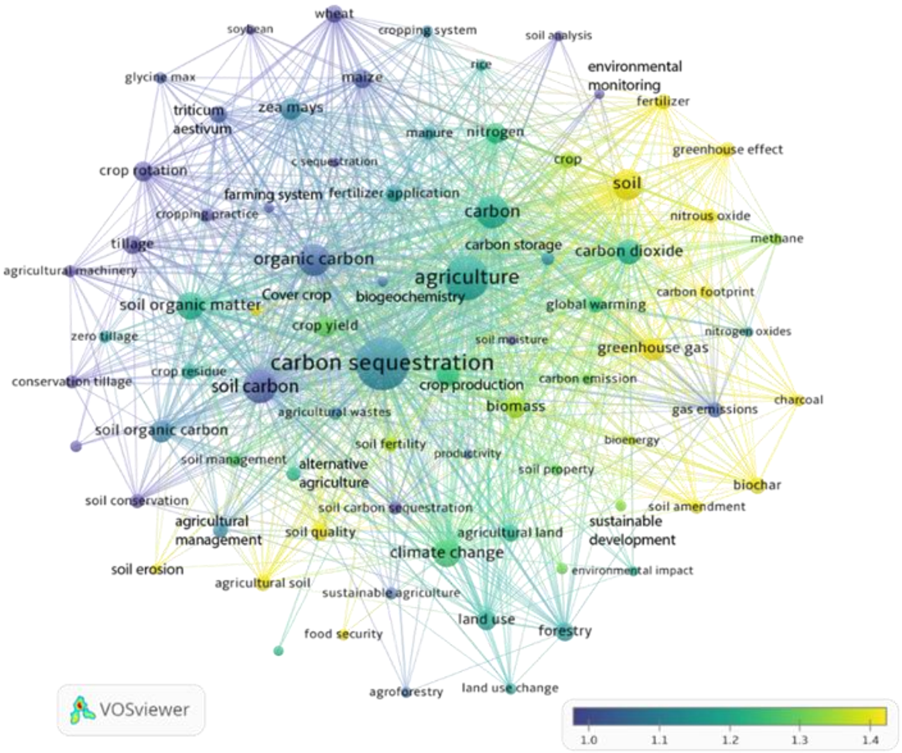

Figure 3-3: Average of citation of bibliometric analysis map of CS publications from 2001-2022..

Figure 3.3 clearly emphasizes the significance of “carbon sequestration” (1033 occurrences and total link strength of 11,190), “crops” (384 occurrences and total link strength of 5276), “agriculture” (770 occurrences and total link strength of 1821), and “soil” (497 occurrences and total link strength of 6754), to increase “soil carbon” (411 occurrences and total link strength of 3339) stock, and then addressing key environmental issues such as “climate change” (352 occurrences and total link strength of 3920), and “global warming” (139 occurrences and total link strength of 1309).

It also highlights the importance of CS, in social strategies like “agricultural wastes” (100 occurrences and total link strength of 1459), “food security” (82 occurrences and total link strength of 798), and “Emissions Control” (77 occurrences and total link strength of 999). Additionally, the figure highlights the significance of the main staple crops in CS, as they are the top three crops found in the figure: maize with 192 occurrences and total link strength of 2897, wheat with 162 occurrences and total link strength of 2486, and rice with 86 occurrences and total link strength of 1384.

3.4 DISCUSSION

The results presented earlier demonstrate the potential to achieve a positive carbon balance, despite the rates being categorized as low to medium. This study provides managers with a threshold for selecting best management practices to reduce emissions and offers policymakers valuable insights for regulation. It underscores the challenges growers face in determining carbon sequestration rates through analytical and modeling methods, highlighting the significant role of economic factors. Additionally, it emphasizes the need to increase the application of data-sharing standards or other cost-effective alternatives.

This study stresses the importance of including co-benefits in sequestration rates to maximize the impact of each tonne of carbon sequestered. Ignoring the values achieved by major staple crops, economic factors, and regulatory policies may escalate the radicality and complexity of its application. This concern aligns with current issues such as soil fertility loss, increased global emissions, reduced sequestration rates, heightened inputs, and the persistent demand for land to sustain the global population. The lack of standardized methods complicates the comparison between practices and hinders progress in the field.

The research needs outlined by this review in this field are as follows:

Developing mechanisms to predict CS potential, it is crucial for growers to make informed decisions. This can be achieved by simplifying the complexity of the process using soil dynamics and software, and applying models aligned with international standards for measuring or calculating CS. Doing so can help to reduce uncertainty, ease verification, and allow for the evolution of the models.

Establishing standards for declaring the CS potential using standardized functional units, soil factors, and boundaries for SOCSR declarations would help the comparison between studies and the evaluation of CS potential among different Carbon Farming practices.

Perform systemic studies, necessary for better understanding of the importance of Carbon Farming in main staple crops for achieving food security, and soil health benefits, as well as for reaching the SDGs and other social strategies associated with soil health. Consider economic incentives and potential negative consequences, such as the accumulation of land by companies or the conversion of food crops to lumber crops, to avoid widening the gap in interpretations.

Create a data quality database to calibrate the actual models in the market, to reduce the time of acceptance and the prices of implementation.

In this systematic review, articles, reviews, book chapters, and conference papers indexed in Scopus from 2001 to 2022 were considered. Specifically, the analysis was limited to long-term experiments, meta-analyses, and reviews reporting an increment in soil carbon stock. The research primarily focused on the CS rates in main staple crops without questioning the methodology used for the estimation but was limited to reporting the estimated values. Also, different practices were analyzed and discussed but without giving a general recommendation of the most effective practice to be chosen but rather giving references for a more informed choice.

3.5 CONCLUSIONS

CS rates in main staple food crops range from very low values to medium values but still show potential to achieve positive carbon balances. This review, performed through an analysis made in VOSviewer, shows that research trends in the field of carbon sequestration are steadily increasing attention and interest. This is mainly due to the topic of climate mitigation potential. CS practices in staple food crops offer significant benefits, including reducing emissions, improving agricultural resilience, and enhancing soil health and food security. However, challenges such as the complexity of CS calculations, the need for standardized methodologies, and the importance of considering cultural and economic factors in crop choices must be addressed. Comprehensive efforts to mitigate and adapt to global warming are essential but must balance practical and cultural agricultural realities to maximize benefits and minimize drawbacks.

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CHAPTER 4

Carbon Farming practices assessment: modelling spatial changes of Soil Organic Carbon in Flanders, Belgium

This chapter is extensively based on the following publication:

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4 CARBON FARMING PRACTICES ASSESSMENT: MODELLING SPATIAL CHANGES OF SOIL ORGANIC CARBON IN FLANDERS, BELGIUM

ABSTRACT

Carbon sequestration in soils is a strategy to mitigate climate change and promote sustainable soil management. Since the European Union (EU) stimulates the reduction of greenhouse gases (GHG) from the atmosphere, the necessity to explore innovative approaches to sequester carbon in agricultural landscapes is becoming urgent. Carbon Farming (CF) has emerged as a promising program to mitigate climate change in agriculture but there is still a lack of agreement on which tools can be used to calculate Soil Organic Carbon (SOC) dynamics in this context. Using the RothC model a spatial analysis of SOC in the agricultural parcels of Flanders, Belgium was performed. Two among the various CF practices were simulated: a use of cover crops (CC) and the most common crop rotations adopted in the area, enriched with the use of cover crops. The performances of the model were evaluated and compared to other studies in areas with similar climate and environments. The selected CF practices can mitigate the carbon emissions from agricultural soils up to 60% of the current projections. The most sensitive variables in the RothC model that affect the final total SOC, and thus determining the model outcome, are the Business As Usual (BAU) carbon inputs and the initial carbon content. For these variables the Pearson Correlation Coefficient with the change in SOC reached values of -0.78 and -0.50 respectively. To achieve net carbon sequestration in the agricultural soils of Flanders, Belgium, more effective solutions need to be evaluated. Furthermore, a larger amount and accessibility of data are required to reach better modelling performances.

4.1 INTRODUCTION

Carbon sequestration in soils is a strategy to mitigate climate change and promote sustainable soil management. Carbon sequestration is related to the removal of carbon dioxide (CO_2) from the atmosphere into long-lasting carbon pools, storing it effectively without risks of leakage (Lal, 2004)). Regarding mitigation solutions in agriculture, the global technical mitigation potential is estimated to be ~5500-6,000 $MtCO_2$-eq/yr, of which the estimated contribution of soil carbon sequestration to the potential is 89% (Smith et al., 2007). The

European Union (EU) is strongly encouraging greenhouse gas (GHG) emission reduction from the agriculture, as the sector accounts for 11% of the European emissions (European Environmental Agency, 2022). The necessity to explore innovative approaches to sequester carbon in agricultural landscapes is becoming urgent. Indeed, in the context of climate change, research in this field in the last 30 years is consistently increasing (Hu et al., 2022).

Carbon sequestration can also provide several benefits for the environment. It can not only mitigate climate change but also have a wide range of other advantages for the environment and the sustainability of agriculture. It can increase the fertility of soils, improve nutrient cycling, maintain biodiversity and ecosystem services, reduce erosion, help microclimate and the management of the hydrological system (Aertsens et al., 2013). Furthermore, with these solutions, it is possible to increase resilience against climate impacts, while contributing to a greater stability of yields and benefit the farm business through more efficient use of crop nutrients and livestock feeding regimes, and diversification of crops (Mcdonald et al., 2021). Therefore, carbon sequestration in agricultural soils is seen as a multidimensional strategy that balances social, economic, and environmental goals. The benefits of carbon sequestration in soils can contribute to the achievement of Sustainable Development Goals (SDGs) in terms of zero hunger, water and sanitation, climate action, and life on land (Bouma et al., 2019; Lal et al., 2018; Paustian et al., 2019b).

In the context of carbon sequestration in soils, Carbon Farming (CF) has emerged as a promising program to mitigate climate change in agriculture and encourage sustainable soil management, whereas conventional agriculture has proven to negatively affect sustainable agriculture (Agovino et al., 2019). CF comprises a variety of agronomic practices that range from land use change to more technological solutions, such as agroforestry, maintaining and improving SOC on mineral soils, managing livestock and manure, and managing nutrients on croplands and grasslands (Mcdonald et al., 2021). The set of Carbon Farming practices involves the management of both land and livestock, of carbon in soils and vegetation, plus fluxes of CO_2, methane (CH_4), and nitrous oxide (N_2O). A subset of these techniques aims at lowering the atmospheric CO_2 content by sequestering it into the soil. Changing agricultural land use and management can help reduce and avoid emissions and sequester carbon in soils and biomass (Keith Paustian, 1998). Some practices such as cover crops, improved rotations, peatland restoration, or expanding agroforestry systems rely on natural processes in agroecosystems (FAO, 2017). Carbon Farming has been subject to growing interest in recent years because agriculture is a sector of fundamental importance to contribute to meeting EU climate goals and because agriculture itself needs to adapt to climate impacts (Anderson et al., 2020). The European Union has been at the forefront of developing comprehensive initiatives and policies to promote CF and reduce greenhouse gases emissions from the agricultural sector by recognizing the importance of carbon sequestration in agricultural practices. The EU's Common Agricultural Policy (CAP), which provides the framework for agricultural support and rural development, has undergone significant revisions in recent years to align with climate and environmental objectives (EC, 2023). The latest CAP reform, covering the period 2023-2027, places a stronger emphasis on environmental sustainability, including measures to enhance carbon sequestration in agricultural soils (Lampkin et al., 2020). As the EU seeks to align its agricultural sector with climate and sustainability goals, the accurate quantification and monitoring of carbon sequestration in agricultural soils becomes imperative. It is important to distinguish the accounting of the mitigation potential of different activities. Each activity has its own co-benefits, trade-offs, and climate mitigation estimates. Climate mitigation can be achieved in the form of C sequestration in soils or in SOC loss mitigation. One of the essential aspects for an appropriate evaluation of such practices is the use of an appropriate terminology when proposing and evaluating these solutions (Don et al., 2023.).

CF includes a wide range of practices, each with different effects on SOC and different feasibility (Bossio et al., 2020; Rumpel et al., 2023). This necessitates the development of robust methodologies, computational models, and monitoring frameworks that can provide estimations to quantify soil carbon sequestration and storage practices (Paustian et al., 2016) and enable the generation of carbon credits. Practices to achieve these ambitious results are reported in several studies (COWI et al., 2021; Mattila et al., 2022; Sharma et al., 2021). The SOC sequestration potential, due to changes in C inputs, can be simulated using dynamic SOC models. In this study, the RothC model (Coleman and Jenkinson, 1996) was employed to assess the carbon sequestration in the topsoil (30 cm) of Flanders, Belgium, under Business as Usual (BAU) practices, a preliminary value essential for the calculation of a baseline, and under the selected sustainable soil management practices. The definition of an accurate baseline is crucial for effective participation in a crediting scheme (Bento et al., 2016). The assessment of a regional scale C sequestration potential for different practices is an important target (Amelung et al., 2020).

For practices such as improved rotations and of the use of cover crops, the spatial variability has not been widely studied yet. Few studies report on the possible trends in C dynamics after the implementation of CF practices (Seitz et al., 2022). This is conducted by evaluating the performances of the model. This study contributes to the ongoing discourse on CF and its integration into carbon markets, as one of the key aspects to achieve a wide adoption of effective CF practices lies in the accurate quantification of the amount of carbon stored in soils. The hypotheses of the research are that CF practices can provide climate mitigation in agricultural soils and that the RothC model can be used as an effective tool to predict SOC changes. The effect of the different model inputs on the results was evaluated to identify the most sensitive model variables and understand where possible actions could be taken. One of the main hypotheses is that the C inputs have a major impact on the simulation. The objectives are to (i) provide a reliable estimation of the annual SOC change for twenty years into the future, which can be used to participate in carbon markets, (ii) model sustainable soil management practices in the framework of CF and their spatial variability, and (iii) evaluate the correlation between the main variables of the RothC model and the simulation results. The results are useful for the implementation of sustainable soil management practices in agriculture. The overall goal of this study is to investigate the effects of selected Carbon Farming practices on the SOC change in agricultural soils of Flanders, Belgium. For this purpose, a spatially explicit version of the RothC-26.3 model was used.

4.2 MATERIALS & METHODS

4.2.1 The RothC Model

RothC-26.3 is a process-based model that allows the simulation of organic carbon dynamics in non-waterlogged topsoils. On a monthly time step, the model simulates changes in five organic carbon pools and in total soil organic carbon. Soil texture, carbon inputs, climate, vegetation cover, and the decomposability of the incoming plant material are the main input data required. Decomposable plant material (DPM), resistant plant material (RPM), microbial mass (BIO), humified organic matter (HUM), and inert organic matter (IOM) are the five organic carbon pools that make up total soil organic carbon (SOC). Incoming plant material has a certain ratio of decomposable plant material (DPM) to resistant plant material (RPM). Most agricultural crops have a default DPM/RPM ratio of 1.44, meaning that the plant material is 59% DPM and 41% RPM. Both DPM and RPM decompose to CO_2, BIO and HUM at specific decomposition rates. In turn, BIO and HUM further decompose to CO_2, BIO and HUM at their own specific rates (Jenkinson and Coleman, 1994; Jenkinson et al., 1997, 1992, 1991, 1987). The decomposition rates depend on climate, clay content and vegetation cover. RothC-26.3 was implemented in R version 4.0.3 using the RothC Model function in the SoilR package (Carlos A. Sierra et al., 2012; R Core Team, 2023). The SoilR default version was modified to include the effects of soil cover. First, the original SoilR moisture function was modified with a "vegetation cover factor" to include the soil cover effect and emulate the original RothC model. Vegetation cover data were also used to determine whether the soil surface was bare or vegetated. NDVI values were rescaled such that 0 indicates bare soil and 1 fully vegetated soil. This modification allows these values to be used as moisture modifying factors and set a threshold of 0.8 to determine if the soil is fully covered by vegetation. The model can be run both in both "inverse" and "forward" modes. The inverse mode allows the calculation of the yearly C input required to reach a given equilibrium SOC content. The forward mode uses predefined monthly C inputs to simulate SOC dynamics.

4.2.2 Input data

The model requires information on climate (monthly air temperature, precipitation and potential evapotranspiration), on soil (clay content and initial SOC content), on land use and on monthly vegetation cover. Climate data, including temperature and precipitation were obtained from the Royal Meteorological Service and processed to monthly level (Gobin and Van De Vyver, 2021; Journée et al., 2019). Potential evapotranspiration was calculated using the Penman-Monteith equation (Allen et al., 1998). The average monthly air temperature, precipitation and potential evapotranspiration were calculated for the periods 1985-2004 and 2005-2020. Data on the initial SOC content in the topsoil (0-30 cm) were obtained from the "databank ondergrond Vlaanderen" webservice. The data are the result of different sampling campaigns conducted in Flanders between 1997 and 2008 with a 40 meters resolution. For the purpose of this study, the SOC map is considered to be representative for the year 2004. To improve the modelling results, pixels that show outliers for the value of SOC in 2004 were removed. Removal is necessary for visualisation purposes. An extremely small number of points would alter the scale of the presentation and would not allow an appreciation in the spatial variability of SOC. The method chosen to select the values to be removed is the interquantile range method: data points with a SOC value higher or lower than 1.5 times the interquartile range (25th percentile) were removed. A limited number of points (6%) was removed. RothC is not recommended for use on organic or organo-mineral soils which are characterized by higher SOC levels (Falloon et al., 2006). Clay information was obtained from the International Soil Reference and Information Centre (ISRIC) SoilGrids 2.0 (Poggio et al., 2021) with a spatial resolution of 250 meters. Data are provided at four different depths and integrated using the trapezoidal rule across soil depths up to 30 cm (Bilas et al., 2022). Land use data were used to determine which pixels fall in the agricultural land class and are obtained from the 2015 European Space Agency dataset (ESA, 2017) with a 300 meters resolution. In this analysis only the areas dedicated to agricultural activities were considered, so the RothC model was run for all pixels classified as “cropland”. Monthly Vegetation Cover (VC) data were retrieved from the MOD13A2 Version 6 product which provides the Normalised Difference Vegetation Index (NDVI) at 1 kilometer (km) spatial resolution (Didan, 2015). Google Earth Engine was used to obtain the satellite information and to calculate the monthly probability of the ground being vegetated (NDVI > 0.6) for the period 2015-2020.

4.2.3 Modelling phases

The modelling phase follows the approach proposed in the GSOCseq Global Soil Organic Carbon Sequestration Potential Map Technical Manual (FAO, 2020). The model was initially run in a inverse mode for a standard 10,000-year timeframe to calculate the yearly C input required to reach the 2004 SOC level, assuming that the SOC level was at equilibrium in that year (“spin-up phase”). The average climate for 1985-2004 was used as input. To reduce the variability of the current soil organic carbon (SOC) stocks at year 0 (representing the initial SOC stocks in 2022) and to account for climatic changes between 2004 and 2022, a secondary initialisation step was implemented ("warm-up phase"). This phase, also referred to as a "short spin-up”, aims to minimize initialization effects (e.g., deviations in the estimation of initial pool sizes). If recent national SOC data had been available this phase wouldn't have been necessary. After a 10,000-year spin-up phase, the model provides estimates of C inputs and of different SOC pools (t Cha-1) in 2004. During the warm-up phase C inputs from 2004 onwards are adjusted according with changes in NPP (FAO, 2020; Smith et al., 2005). The NPP was calculated using the consolidated MIAMI model (Lieth, 1975) and was set equal to the average annual NPP over the period 1985-2004 for the first modelling year in the warm-up phase. Finally, the model was run in the forward phase to simulate SOC dynamics for the next twenty years (“forward phase”). A 20-year timeframe was chosen because the crediting period for Carbon Farming mechanisms varies from 5 to 20 years (Commission et al., 2021) and SOC requires several decades to reach a new equilibrium.

Again, the average climate for the period 2005-2020 was used, assuming that the climatic conditions will not change significantly over the next twenty years.

4.2.4 The modelling scenarios

The model scenarios analysed relate to BAU conditions and two different alternative scenarios to enrich soils with organic carbon: improved rotations and the use of cover crops. These are well-known practices for climate mitigation in agriculture (Bumbiere et al., 2022). Cultivating crops and varieties that produce a dense foliage, a deep root system and leave a high amount of biomass on the field can be beneficial for the soil C balance. The annual C inputs used in the forward phase were adjusted according to the scenario. First, the BAU conditions were studied, using the C inputs obtained in the spin-up phase and adjusted in the warm-up phase. The BAU C inputs were specific to each pixel and unchanged in each year, as no climate change and land use change was assumed. Subsequently CF scenario simulations were performed. Four different crop rotations, which are among the most common in Belgium, and the use of cover crops (yellow mustard, Sinapis alba) were analysed to understand their effects on SOC stocks of Flemish soils. The choice to study separately improved rotations and the use of cover crops is based on the fact that the aim is to assess the outcome of different management practices on conventional scenarios (improved rotations, evaluation of the most common rotations + cover crops) and on the ensemble of agricultural soils of Northern Belgium (cover crops all over Flanders as an improvement of BAU scenario).

In the first analysis, four crop rotations (**Table 4.1**) were analysed. Each rotation is improved each year with the use of yellow mustard as a cover crop. The primary driver of the increase in soil organic carbon following the adoption of cover crops is attributed to the increased C input besides other minor effects (Bolinder et al., 2020) which were therefore not taken into account. Carbon inputs were adjusted to account for an additional C input to the soil provided from the use of cover crops (Seitz et al. 2022). To run the forward phase, the annual C inputs and the decomposability (DPM/RPM ratio) of the incoming crop material is adapted to the specific rotation scenario. Crop specific C inputs and DPM/RPM ratios were obtained from the study carried out by the Soil Service of Belgium and Ghent University (Soil Service of Belgium, Ghent University, 2006). DPM/RPM ratios of common crop residues after incorporation into the soil were taken according to specific analysis conducted in Belgium as well as C inputs values. Data analysed were verified from national productivity statistics. The C input was determined from fresh biomass and an average C%

per crop type, while the DPM/RPM ratios were calculated as a weighted average including the proportion of roots and above-ground plant residue. An average annual C input and DPM/RPM were calculated depending on the crop type and frequency of occurrence in the rotation. When winter cereals occurred in the rotations, the contribution of cover crops was not considered because the possibility of overlapping cultivations was not taken into account. To represent the spatial variability of crop productivity in Flanders, a conversion factor for the C inputs was applied. The conversion factor for a specific pixel is given by the ratio between the NPP of that point and the mean NPP for Flanders, where the NPP is calculated using the MIAMI model.

Table 4-1: Specific crops belonging to each improved rotation scenario, yearly C input and decomposability (DPM/RPM ratio).

Rotation Scenarios	**Crops**	**Carbon input (tC ha^{-1} year^{-1})**	**DPM/RPM**
Rotation 1	Grain maize (Zea mays) monoculture	4.56	1.36
Rotation 2	Silage maize (Zea mays) monoculture	2.61	1.32
Rotation 3	1 year of silage maize and 2 years of winter cereals (usually Triticum aestivum)	3.21	1.33
Rotation 4	2 years for winter cereals, 1 year of silage maize and 1 year of potatoes (Solamum tuberosum)	3.19	1.37

In the second analysis, the use of cover crops was considered as an improvement to the BAU scenario. For the business-as-usual scenario, annual C input is taken from the warm-up phase. There are no significant previous studies on the potential of cover crops to increase C inputs in northern Belgium in relation to the actual conditions. Some previous studies investigated the effects and the implementation of cover crops in Belgium (Dendoncker et al., 2004; Van De Vreken et al., 2016). Another analysis showed results of simulations for a neighbouring region, west Germany, in which the C inputs increase from cover crops is of 12% with respect to the BAU (Seitz et al., 2023). Given the lack of accurate data, in the present study cover crops are assumed to supply an additional 12% C input to each point of the area of interest, with respect to the BAU scenario. The DPM/RPM ratio used in this case is the default value of 1.44 for agricultural land (Coleman and Jenkinson, 1996).

In this research, the SOC sequestration potential or SOC accumulation potential [tC ha-1] is the difference between the simulated SOC stocks [tC ha-1] of each simulated scenario and the simulated SOC stock in BAU scenario [tC ha-1]. If the difference is negative, there is SOC loss instead of SOC accumulation [tC ha-1].

The simulation is conducted for all agricultural arable land, already used for annual crops, in Flanders (Belgium). The adoption of these practices is assessed in a “what if” scenario: the aim of this work is to understand how SOC would change if the proposed techniques were applied in land that is already arable and would not require land use conversion. The results are presented as graphs and maps. Graphs represent the change in SOC stocks from year to year in each scenario, with each year representing the median value of SOC stocks in Flanders. The median value was chosen because it was considered more appropriate to show the central tendency of the simulated values, since a single value could be used to build such graphs. Instead, maps are useful to show the spatial variability of the simulated carbon content in each scenario and provide an overview of the change in SOC sequestration potential in the study area.

4.2.5 Statistical analysis

The performance of the RothC model was evaluated using a validation dataset consisting of SOC measurements from the Land Use and Coverage Area frame Survey (LUCAS) carried out in 2009, 2015 and 2018 (European Commission. Joint Research Centre., 2020; European Commission. Joint Research Centre. Institute for Environment and Sustainability., 2013; Fernandez et al., 2022). For specific points in Flanders, the Organic Carbon (OC) content [gkg^{-1}] was available from the LUCAS database. Values of SOC stock [$tCha^{-1}$] were obtained using the bulk density estimates [kg m-3], corrected for the presence of coarse fragments (Poeplau et al., 2016), associated to the respective OC values and the thickness of the considered soil layer (30 cm). For all the validation data points, the warm-up phase was run from 2005 to 2009, 2015 and 2018, and the obtained simulation results were compared with the measured SOC in the respective year. The validation was conducted by means of a correlation analysis. The regression curve was built in terms of ∆SOC. This is the difference between the final SOC value and the initial SOC value in 2004. In this phase, the simulated values of ∆SOC were compared with the measured values. ∆SOC was chosen because it was considered to be more representative of the behaviour of the model. It allows both SOC gain and SOC loss to be represented, since one of the main aims is to understand changes in SOC stocks. The performance of the model was evaluated for each year and considering all years together (aggregate). The selected statistical indicators are the coefficient of determination (R2), the Mean Absolute Error (MAE), the Root Mean Squared Error (RMSE) and the Cohen's d (d).

Finally, the correlation between the results and the main variables of the RothC model is presented. The influence of the main variables on the final values obtained through the simulation process was assessed by statistical correlation between the values, in the form of the Pearson Correlation Coefficient (PCC). The selected main variables analysed are clay content, SOC content at the start of the simulation (referred to as SOC 2022), C inputs, temperature and precipitation. The output value is ∆SOC, to which the correlation with the input variables is calculated. Statistical analysis was performed in the R environment with Hmisc and HydroGOF package(Harrell Jr F, 2023; Mauricio Zambrano-Bigiarini, 2020) to calculate the indicators, and the coefficients of the correlation.

4.3 RESULTS

4.3.1 SOC Trends

The annual trend, taken as the median value of the simulation for Flanders, shows a decrease in SOC stocks under BAU conditions. Under the current conditions, agricultural soils in Flanders are net carbon emitters (**Figure 4.1**). Each scenario leads to a decreasing trend in SOC stocks, both with the use of cover crops (CC) and with the main crop rotations in Belgium, enriched with yellow mustard.

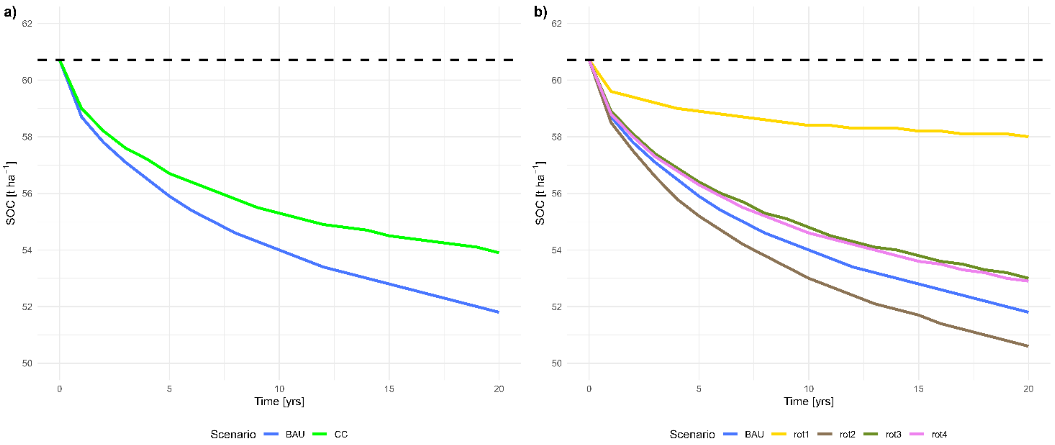

Figure 4-1: SOC trends under different management scenarios: a) Use of cover crops (CC) compared to the BAU scenario; b) improved rotations1, 2, 3 and 4 compared to the BAU scenario. See Table 4.1 for the crops in rotation.

In the BAU scenario, agricultural soils in Flanders will release up to 9 tCha^{-1} in twenty years from now. As expected, the use of CC mitigates the loss observed in the BAU scenario, but a net loss of C can still be predicted. The implementation of CC leads to a loss of 6.5 tCha^{-1} in twenty years from now. Due to the higher C input into the soil from grain maize, rotation 1 results in a slower rate of decline than the other rotations. This results in a net loss of 2.5 tCha^{-1} C twenty years from now. This value is significantly lower than for the other rotations where it reaches 10.5 tCha^{-1}, 7.5 tCha^{-1}, 7.5 tCha^{-1} for rotation 2, rotation 3 and rotation 4 respectively.

4.3.2 Spatial variability

The SOC sequestration potential in terms of spatial variability is reported at the end of the modelling phase, after 20 years, and shows a relative change compared to the BAU (**Figure 4.2**). The SOC changes indicated in the maps should therefore be understood as the difference between the SOC content in the specific scenario and the SOC content in the BAU scenario, at the end of the modelling phase (i.e. year 20). In the maps, SOC accumulation fades from white to green, whereas SOC loss fades from white to red. The use of CC can provide an increase in SOC stocks of 3 tha^{-1}, homogeneously distributed throughout Flanders. Improved crop rotation 1 can also provide an evenly distributed increase in SOC stocks of up to 8.5 tCha^{-1}. In contrast to the previously presented scenarios, improved rotation 2, 3 and 4 show spatial variations of the SOC accumulation potential. Rotation 2 exhibits a marked SOC loss, especially in the central zones of the region, corresponding to the decreasing trend observed in **Figure 4.1**. SOC losses reach up to $^{-1}$0 tCha^{-1}. In this scenario, only the southern part of the study area shows a modest SOC accumulation of up to 3 tCha^{-1}. Not surprisingly, given the results got from the annual trend, the improved rotations 3 and 4 also show similar results in terms of spatial variability. Both rotations are characterized by a higher SOC accumulation potential in the southern region, up to 6 tCha^{-1}. These two scenarios display a substantial SOC loss in the central part of the study area, up to a maximum of -5 tCha^{-1}.

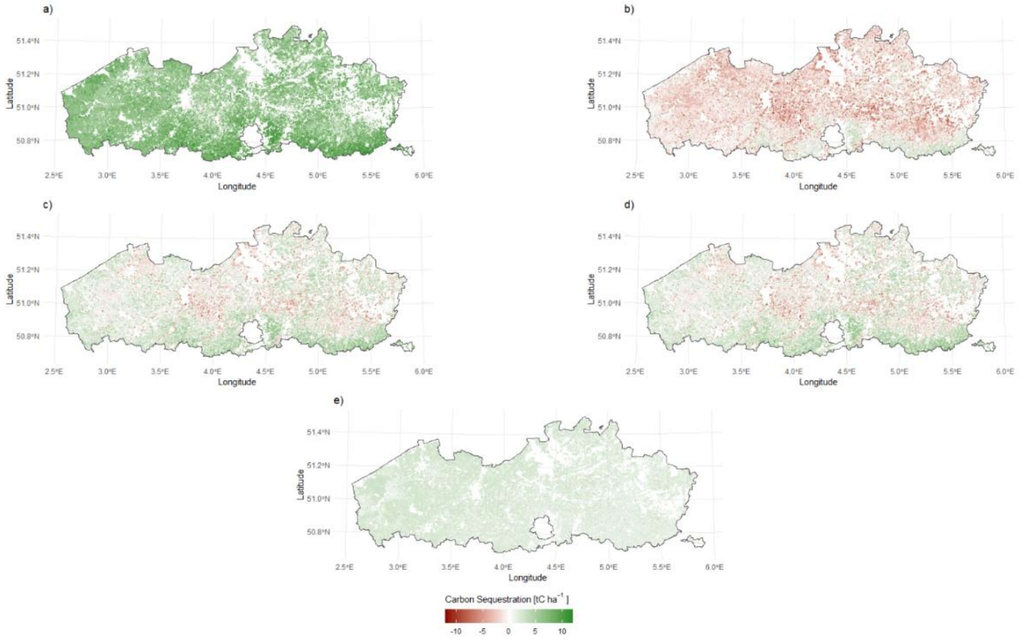

Figure 4-2:SOC sequestration potential at twenty years with respect to the BAU (relative change): a) Improved rotation 1; b) improved rotation 2; c) Improved rotation 3; d) improved rotation 4; e) cover crops

Looking at the SOC stock in 2022, its spatial variability across Flanders is large (**Figure 4.3.a**). The SOC stock is higher in the north-west, mainly due to the presence of higher clay content. It also peaks in the central and eastern part of the region, reaching values of 90 tCha^{-1}. The lowest values of SOC content are observed in the southern part of the region, from west to east, with values not exceeding 50 tCha^{-1}. The BAU C inputs calculated in the warm-up phase are represented in terms of spatial distribution (**Figure 4.3.b**). C inputs at equilibrium, in 2022, range from a minimum of 1.37 to a maximum of 7.36 $tCha^{-1}yr^{-1}$. C inputs were not evenly distributed over the area. C inputs peak in the central and the eastern parts of the study area. The lowest values are again found in the southern part, from west to east.

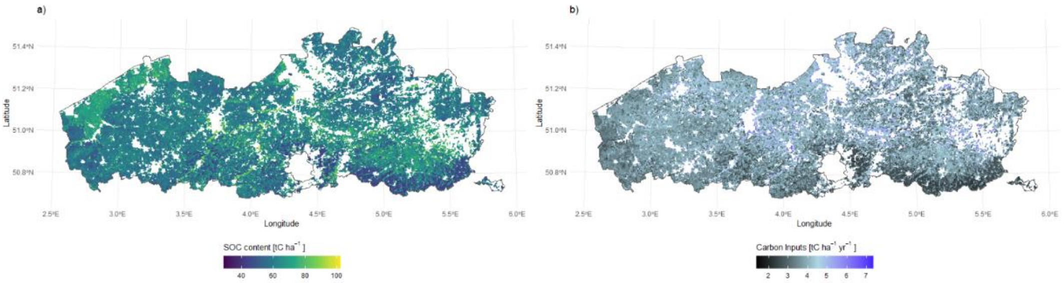

Figure 4-3: a) SOC stock in the upper 30 cm soil for the year 2022, values reached from the warm-up phase; b) Spatial distribution of annual C inputs used to simulate the BAU scenario.

4.3.3 Statistical analysis

4.3.3.1 Validation of the study

The relationship between the measured and simulated ∆SOC for the years 2009, 2015 and 2018 resulted in an overall R^2 of 52% (**Figure 4.4**).

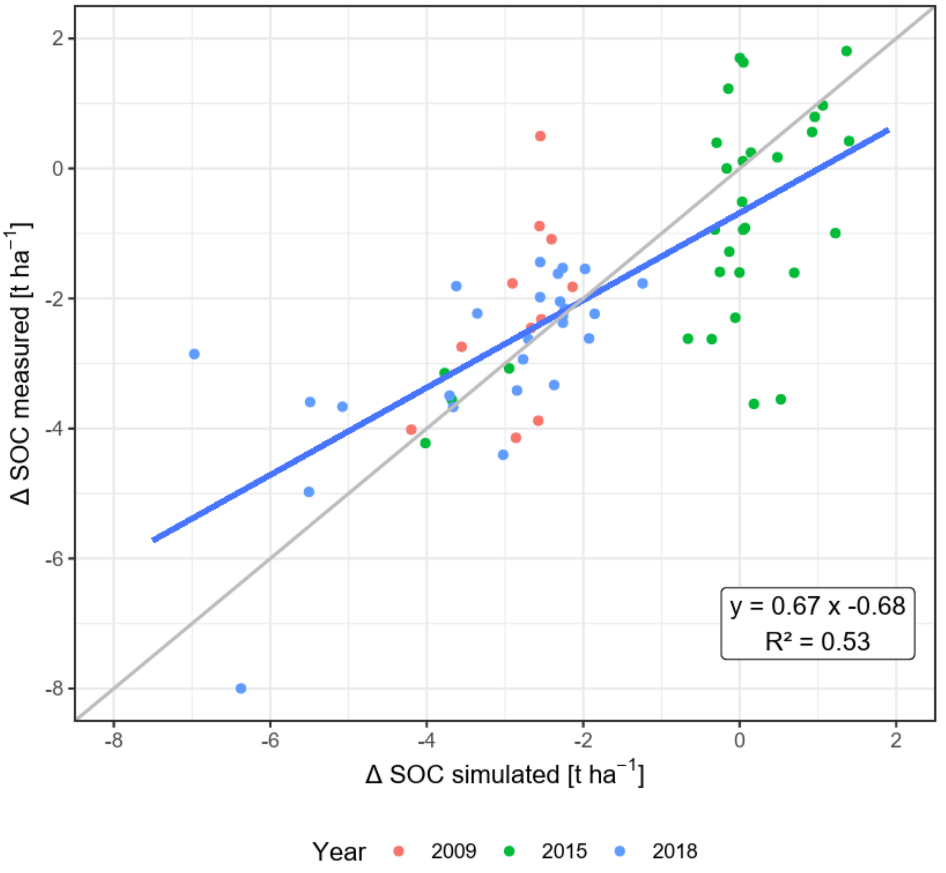

Figure 4-4: Regression analysis of measured against simulated values of ΔSOC, the difference between final SOC value (2009, 2015 or 2018) and starting SOC value (2004).

Table 4.2 shows the coefficient of determination (R2), Mean Absolute Error (MAE), Root Mean Squared Error (RMSE) and Cohen's d (d). The statistical correlation (R2) of individual years is weaker than that of the aggregate. The year 2018 has lower values in terms of MAE and RMSE than all the other years, hence it shows a better performance. The effect size is large for each year and for the aggregate values, except for the year 2009, which has the lowest value of d.

Table 4-2: The performance of the RothC model simulation is evaluated in terms of R2, MAE, RMSE and d.

Year	Number of observations	R^2	MAE [$tCha^{-1}$]	RMSE [$tCha^{-1}$]	d [$tCha^{-1}$]
2009	11	0.20	0.92	1.18	0.64
2015	31	0.52	0.92	1.33	0.75
2018	26	0.40	0.72	1.03	0.75
Aggregate	68	0.53	0.93	1.22	0.83

4.3.3.2 Correlation among the results and the main input variables of the RothC model

For the BAU scenario, ∆SOC is mainly correlated with C inputs and with SOC in 2022 (**Figure 4.5**). Temperature shows a weak inverse relationship with ∆SOC, while precipitation has an even weaker direct relationship with ∆SOC. As expected, increasing clay content is positively correlated with SOC changes. Concerning the CC scenario, the correlation with C inputs is direct, changing direction with respect to the BAU scenario. The improved rotations display a similar behaviour in terms of the correlation between the final result and the main input variables. A strong negative relationship can be observed between the ∆SOC and C inputs. Precipitation and temperature have a weak negative influence on the results, this is only valid for the improved rotations.

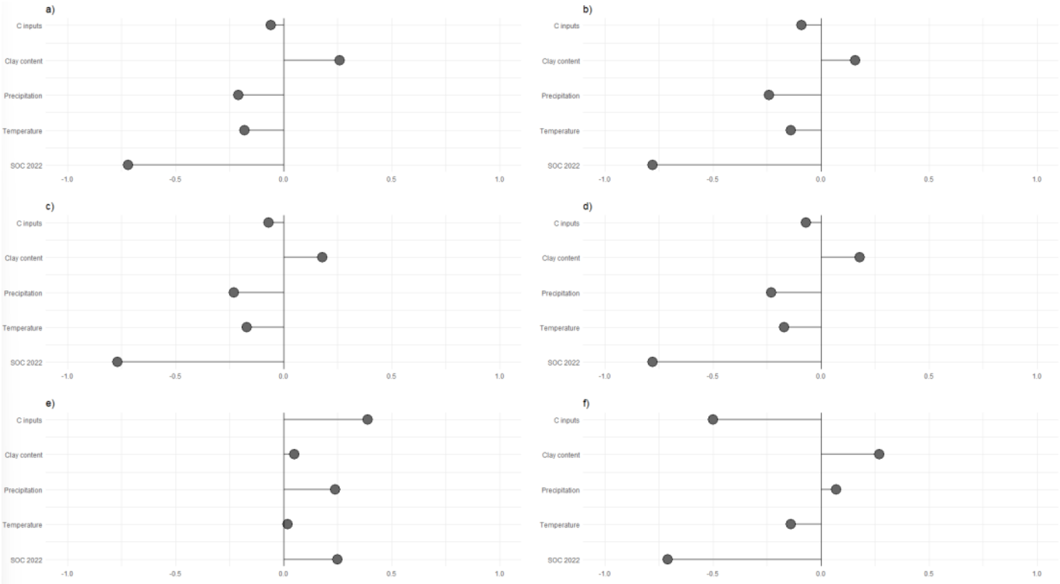

Figure 4-5: Correlation between ∆SOC and RothC main input variables for the six different scenarios, evaluated in terms of the Pearson Correlation Coefficient. a) Improved rotation 1; b) improved rotation 2; c) improved rotation 3; d) improved rotation 4; e) cover crops; f) BAU

4.4 DISCUSSION

The RothC model was used to predict SOC stock changes in the agricultural soils of Flanders. The resulting trends showed that the agricultural soils of this region can be considered as net C emitters, especially in the BAU scenario. This phenomenon has already been widely observed (Lal, 2023) and is not unique to the soils of Flanders. The use of cover crops (CC) provides a mitigation effect by reducing SOC loss by ~20%. The simulated trend (**Figure 4.1-a**) can be compared with the result obtained by (Seitz et al., 2023), who evaluated the effects of CC on SOC in western Germany. However, they modelled the trend after 50 years, whereas the simulations of this study stopped after 20 years, but similar trends can be observed. In their analysis, the decreasing trend stopped after the 20th year and then the trend started to increase. As already mentioned, the simulations were stopped at 20 years because of the current structure of carbon markets. No previous studies have shown the annual SOC change associated with improved rotations. However, the results seem to be as expected: improved rotation 1 (grain maize monoculture), characterized by the highest C inputs, is by far the most effective rotation in reducing SOC losses. Rotation 3 (silage maize and winter cereals) and rotation 4 (silage maize, winter cereals and potatoes) show a similar behaviour, mainly driven by the low C input, slightly slowing down the SOC loss process. Rotation 2 (silage maize monoculture), having the lowest C inputs in the scenario, is the only scenario which increases this loss. The main difference between grain maize and silage maize is that in silage maize almost the whole plant is harvested (for animal feed), whereas in grain maize much more crop residues are left on the field. More crop residue left on the field is beneficial for SOC accumulation (Gamble et al., 2021).

4.4.1 Spatial variability

Looking at the spatial variability maps (**Figure 4.2**), the same trends discussed in section 3.1. can be understood. CC implementation (**Figure 4.2-e**) shows an overall, uniformly distributed SOC accumulation across the region. This is due to the evenly distributed decomposability of the incoming plant material and primarily to the same percentage increase in C inputs. From **Figure 4.2-a** it is notable that improved rotation 1 would be highly effective throughout the analysed area. This is not the case for the other improved rotation. Again, rotation 3 and rotation 4 show a comparable development among themselves (**Figure 4.2-c, Figure 4.2-d**). The central area of the region is characterized by SOC loss. Therefore, it is important then to understand the factors that determine the opposite outcome with respect to the rest of the area. The main variable affecting this kind of performance is related to the BAU C inputs (**Figure 4.3**). This also explains the values obtained from the simulation of the improved rotation 2. Areas with higher BAU C inputs (central and eastern parts of Flanders) are less responsive to SOC accumulation and in some cases also prone to SOC loss. These areas require a higher C supply to maintain current SOC stocks and would therefore require even higher C inputs to increase SOC. In contrast, areas with lower BAU C inputs (the whole southern part of the region) can maintain current SOC stocks with a modest C supply and are therefore suitable for SOC accumulation. The evidence for the relationship between C inputs and changes in SOC has been already discussed in previous studies (Smith, 2004). In the framework of Carbon Farming, the quantification of SOC changes, especially with respect to spatial heterogeneities, is one of the essential requirements for the implementation of soil carbon certificates as a suitable tool for climate mitigation (Paul et al., 2023).

4.4.2 Correlation

The correlation indices (**Figure 4.5**) highlight the qualitatively observed results of the spatial analysis. In the BAU scenario, the strong negative correlation of ∆SOC with initial SOC (which in **Figure 4.5** is shortened as “SOC 2022”) and with C inputs confirms the previous explanation. In every scenario, the negative correlation between ∆SOC and initial SOC provides further evidence: where initial SOC content is higher, greater efforts should be made to increase SOC stocks by increasing C inputs. Furthermore, it is interesting to investigate the effect of the other main input variables on the results in the study area. As expected, the clay content has a positive correlation with the ∆SOC due to the chemical properties of clay minerals to bind with Soil Organic Matter (SOM) (Churchman et al., 2014; Singh et al., 2018). In fact, the SOC stock is higher in the north-west of Flanders, close to the coast, because of the presence of a higher clay content in this part of the region. However, a positive correlation of SOC with Fe and Al concentrations in the silt and clay fraction suggests enhanced carbon sequestration due to the formation organo-metal complexes (Van De Vreken et al., 2016). Temperature has a slight negative correlation with changes in SOC stocks. Temperature has complex effects on SOC stocks that depend on several variables, including soil moisture, plant biomass output, microbial activity and a number of environmental constraints (Dash et al., 2019). Some research indicates that SOC loss rises with temperature (Ghimire et al., 2019), but other research indicates that SOC inputs and management strategies are more crucial for maintaining SOC stability in warmer environments (Yu et al., 2022). In the agricultural soils of Flanders, the trend suggests a SOC loss with increasing temperatures, irrespective of the change in management which mainly causes changes in decomposability and C inputs. In the BAU and CC scenarios, precipitation has a modest positive correlation with ∆SOC. The correlation changes with changing the management practices, because the correlation is negative in the improved rotation scenarios. The effect of precipitation on SOC stocks is complex and depends on various factors such as ecosystem type, management practices, and soil moisture. Studies suggest that the relationship should be positive, as a reduction in precipitation can have a negative effect on SOC stocks in agriculture (Puche et al., 2023).

4.4.3 Validation

The validation phase of SOC models and the use of appropriate datasets has already been discussed in the literature (Le Noë et al., 2023). In the present study, the validation was conducted considering aggregated values produced results in line with previous findings for areas with similar climate and morphology (Dechow et al., 2019). Research conducted in the western part of Germany obtained R^2 values for the comparison of simulated and measured ΔC, ranging from 0.45 to 0.52 across different datasets, whereas in this study the obtained R^2 value is 0.52. A major limitation is the limited availability of validation data. The availability of a topsoil (0-30 cm) database such as the LUCAS is an important resource, but a broader set of information would be necessary to perform a more accurate analysis. Remotely sensed imagery need further research before being suitable for the purpose since predictions errors are still large (Orton et al., 2023). Secondly, the RothC Model is not able to simulate the management practices directly. The only way to account for their implementation is to incorporate their effects on crop residue and it is often challenging to select relevant information (Nemo et al., 2017). Furthermore, the RothC model does not account for topography. The topography of Flanders, Belgium, is characterized by a relatively flat landscape with some low-lying hills and a sedimentary origin, characterized in particular by loamy soils that are prone to erosion and sediment transports (Meersmans et al., 2008; Van Oost et al., 2007). Topography can indirectly affect SOC dynamics by influencing soil erosion, water movement, and soil organic matter distribution. At a specific site, soil erosion can have a significant impact on C stocks. However, some studies suggest that at broader scales, erosion may actually reflect a redistribution of soil C rather than a loss (Paustian et al., 2019a). Indeed, lateral fluxes play an important role in SOC dynamics (Nadeu et al., 2015). Despite all of this, the model has proved to perform well in simulating SOC dynamics in agricultural soils of Flanders, Belgium.

4.5 CONCLUSIONS

This research addressed the potential suitability and effectiveness of practices aimed at increasing C sequestration in agricultural soils. The analysed practices showed a potential to reduce soil C emissions in Flanders, Belgium. However, the selected sustainable soil management practices are not able to reverse the trends and turn the agricultural soils of Flanders into net sinks. The studied crop cover and improved rotations have the potential to reduce soil C emissions. The RothC model was used for the simulation and led to the conclusion that that, given the pedoclimatic conditions of the region, the main influencing factors are the initial C stocks and the BAU C inputs. Higher initial SOC levels required higher C inputs to maintain the same SOC level. The same areas were more prone to SOC loss, with little influence from clay content or climatic conditions. Our findings confirmed the possibility to use RothC as a model for predicting

SOC changes in Flanders, Belgium. RothC can be used to simulate SOC dynamics of different management practices, if the appropriate data are available. Future research should focus on overcoming the current limitations in data availability, especially concerning the C related inputs. More complete approaches will also rely on measured data from laboratory experiments and field measurements in the study area. Concerning Carbon Farming, this study presents an overview of the possible impacts of selected practices on soils carbon stocks in Flanders. Most carbon sequestration projects aim to sequester "new" carbon by enrolling producers who have not previously used eligible methods. This implies that producers who have previously used eligible methods could not qualify for these initiatives.

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CHAPTER 5

From Soil Carbon towards System Sustainability: Integrating SOC Modelling and Life Cycle Assessment to evaluate environmental trade-offs in Carbon Farming

This chapter is extensively based on the following publication:

Spotorno, S., Gobin, A., Vazquez, D. A. A., Gagliano, E., Del Borghi, A., & Gallo, M. (2025). *From Soil Carbon towards System Sustainability: Integrating SOC Modelling and Life Cycle Assessment to evaluate environmental trade-offs in Carbon Farming;* ***https://doi.org/10.1016/j.farsys.2025.100195***

5 FROM SOIL CARBON TOWARDS SYSTEM SUSTAINABILITY: INTEGRATING SOC MODELLING AND LIFE CYCLE ASSESSMENT TO EVALUATE ENVIRONMENTAL TRADE-OFFS IN CARBON FARMING

ABSTRACT

The European Union (EU) is committed to reducing greenhouse gas (GHG) emissions to the atmosphere and promoting sustainable soil and land management practices. Carbon Farming (CF) is a set of practices to mitigate climate change in agriculture through carbon sequestration in soils. While CF practices increase soil organic carbon (SOC) stocks, they are also expected to have environmental impacts and potential trade-offs. However, the environmental impact of CF practices is often overlooked, and a comprehensive evaluation using Life Cycle Assessment (LCA) methodology is required. The RothC model was used to simulate SOC dynamics under different CF practices on arable land in Northern Italy: reduced tillage (RT), farmyard manure (FYM) application, and cover crops (CC). LCA methodology was applied to quantify GHG emissions and other environmental impacts beyond carbon. The results confirm that soil management strategies significantly influence SOC accumulation, but their environmental trade-offs differ substantially. FYM application sequesters the most carbon (4.89 t C ha^{-1} over 20 years) due to exogenous carbon inputs. RT (1.34 t C ha^{-1}) and CC (1.73 t C ha^{-1}) also contribute to sequestration, but at lower rates. However, LCA results revealed significant trade-offs: while FYM maximizes carbon sequestration (0.90 t CO_2 ha^{-1} yr^{-1}), it substantially increases acidification (+254%), marine eutrophication (+372%), terrestrial eutrophication (+243%), and photochemical ozone formation (+290%) compared to conventional agriculture. In contrast, CC and RT provide a balanced profile with moderate sequestration benefits (0.32 and 0.25 t CO_2 ha^{-1} yr^{-1}, respectively) and reduced environmental impacts, with RT showing improvements across all acidification and eutrophication indicators. This research underlines the critical need for comprehensive system assessment of agricultural sustainability, as CF may place too much emphasis on carbon sequestration without fully considering other environmental impacts requiring mitigation.

5.1 INTRODUCTION

The growing need to mitigate climate change has intensified efforts across all productive sectors, with agriculture receiving particular attention for its potential role in soil carbon sequestration (Nazir et al., 2024). Carbon sequestration in soils refers to the long-term storage of carbon in soil organic carbon (SOC) pools, helping to lower atmospheric concentrations of greenhouse gases (GHG) and support sustainable land management techniques (Lal, 2015). Because of its positive role in carbon sequestration, agriculture has a significant mitigation potential. In the European Union (EU), the land use, land-use change, and forestry (LULUCF) sector reported negative values (-7% in CO_2-eq) (EEA, 2022). In this context, Carbon Farming (CF) is a promising strategy, as it involves practices primarily aimed at carbon sequestration, which is often measured in terms of SOC gains and greenhouse gas (GHG) balances (Petersson et al., 2025), as well as improving soil health and biodiversity (Khangura et al., 2023). Carbon Farming refers to agricultural practices that increase the sequestration and storage of carbon in soils and biomass, while reducing soil-related GHG emissions; it partially overlaps with regenerative agriculture, in terms of management practices, but is distinguished by its explicit and quantified focus on carbon removal. Unlike conventional farming, which tends to lead to soil degradation and GHG emissions, CF practices aim to restore soil systems and increase SOC stocks by using cover cropping, improved crop rotations, agroforestry, no-till and reduced tillage farming, organic farming and livestock integration (Schreefel et al., 2020). These practices not only sequester carbon but improve soil fertility and water retention capacity, and increase resilience to climate stress, in line with several Sustainable Development Goals (SDGs), such as zero hunger, climate action, and life on land (Lal et al., 2018; Bouma et al., 2019).

Despite evidence of the potential benefits under different soil management systems, actual carbon sequestration remains limited or inconsistent because of the barriers to the adoption of CF practices, especially in the Mediterranean region. Key barriers include limited economic incentives and high initial costs, lack of access to specialized equipment and technical knowledge, ineffective information dissemination in rural areas, and misalignment with traditional farming practices, fostering skepticism among farmers (Gonzales-Gemio and Sanz-Martín, 2025). Nevertheless, there is a significant opportunity to improve soil health by increasing the adoption of appropriate management practices (Heller et al., 2024).

Research into carbon sequestration in agricultural soils is steadily increasing, mainly due to the climate mitigation potential of CF practices (Arellano Vazquez et al., 2024). These practices show promising SOC gains and potential GHG mitigation. However, robust computational methods and techniques are needed to accurately determine the carbon sequestration potential of CF practices and their eventual environmental trade-offs. Existing studies typically focus on either carbon sequestration potential (Li et al., 2025) or the environmental impacts of agricultural practices, paying particular attention to Global Warming Potential (GWP) (Shi et al., 2026). While some studies have coupled carbon sequestration potential with Life Cycle Assessment (LCA), these are mostly limited to climate indicators (e.g. GWP) and do not explore a broader set of impact categories or environmental trade-offs (Goglio et al., 2014; Fantin et al., 2022). Currently, integrated assessments that couple process based SOC modelling with LCA are lacking. In particular, few studies have combined process based SOC modelling and LCA across multiple impact categories for CF practices in Mediterranean arable systems.

This study contributes to understanding the implementation of CF practices by combining a process-based soil model with LCA to quantify SOC dynamics, carbon sequestration and multi-impact environmental trade-offs. Soil models such as the RothC model (Coleman and Jenkinson, 1996) can simulate various land use and management scenarios using a small number of input parameters (Spotorno et al., 2024). LCA, on the other hand, is a systematic approach to evaluating the environmental benefits and costs of alternative farming methods to those traditionally used (Finnveden et al., 2009).

The novelty of this research lies in its evaluation of the broader environmental impacts of CF, extending beyond carbon sequestration. Integrating the RothC model with the LCA methodology enables a more comprehensive evaluation, ensuring that methods which encourage carbon sequestration do not inadvertently lead to increased environmental impacts in other areas (Brandão et al., 2013).

For the purpose of this study, the selected practices consist of implementing cover crops, transitioning from mineral/synthetic fertilization to organic fertilization and reducing tillage intensity. The main objectives of this study are to (i) estimate the carbon sequestration potential of the selected CF practices, (ii) compare their environmental impacts using LCA methodology, and (iii) identify optimal and sustainable strategies for increasing SOC sequestration while mitigating climate change. The final aim is to evaluate the carbon balance (CB) of a conventional three-year crop rotation in northern Italy, with and without

CF practices. The CB is calculated as the difference between the GHG emissions associated with the agricultural phase of crop production and changes in SOC stock.

This study provides information on the environmental impact of crop production and helps assess management strategies that can improve agricultural sustainability. The findings are also relevant to the EU Carbon Removal Certification Framework (CRCF), which seeks to establish a reliable mechanism for monitoring, reporting, and certifying carbon removals. This research contributes to the development of science-based criteria for certifying soil-based carbon removals that are aligned with the EU's climate and sustainability goals by providing robust data on SOC dynamics and their environmental impacts. By identifying CF practices that optimise carbon sequestration while minimising environmental trade-offs, this research contributes to a better definition of Voluntary Carbon Markets (Kochar et al., 2025) and supports the transition towards sustainable agricultural systems.

5.2 MATERIALS & METHODS

5.2.1 Study area and scenario definition

The study was conducted on a subset of twenty farms from the HelpSoil project, a demonstration project in the Po Valley (Italy) that implemented conservation agriculture on 20 farms over three cropping seasons to assess agronomic, environmental, and economic effects versus conventional systems (Perego et al., 2019). Primary management and input data (tillage intensity, fertilization rates, crop yields, machinery use) were collected from the HelpSoil project farm surveys and field records to ensure site-specific representation of the Po Valley region. Four of the twenty farms from the HelpSoil project were excluded from this study because they were not suitable for the purpose (e.g. due to rice cultivation - waterlogged conditions), or because these farms lacked key information to perform the soil organic carbon (SOC) modelling and Life Cycle Assessment. The assessment focused on a three-year crop rotation (maize - soya - wheat) and compared conventional systems with Carbon Farming (CF) practices.

Four main scenarios (**Table 5.1**) were defined to evaluate SOC dynamics and associated greenhouse gas (GHG) emissions: Business-As-Usual (BAU), Cover Crops (CC), Reduced Tillage (RT) and the transition from inorganic fertilizers to organic fertilizers - FarmYard manure (FYM).

The BAU scenario reflects the conventional agriculture methods used in the Po Valley, which include heavy tillage, a reliance on mineral fertilizers, and a lack of cover crops. Crop residues are the main source of carbon input. This situation served as the baseline against which Carbon Farming (CF) practices were evaluated. The CC scenario assumes the introduction of cover crops between cash crop cycles, thereby increasing annual carbon inputs to the soil and modifying soil cover. However, additional field operations increased diesel consumption, e.g. sowing, mowing, and biomass incorporation, and contributed to indirect emissions. Other inputs (e.g., pesticides, fertilizers) remain unchanged from the BAU scenario. The RT scenario models the adoption of reduced tillage practices, which reduce soil disturbance and slow the decomposition of organic matter. Scenario emissions include reduced diesel consumption, with all other inputs remaining constant. In line with empirical evidence, the RT scenario was also associated with yield penalties (see Section 2.2 for details), which were included to reflect realistic agronomic trade-offs. In the FYM scenario, mineral fertilizers were reduced by 60% and replaced with farmyard manure to maintain N-P-K nutrient balance and increase exogenous carbon inputs. To maintain stable N-P-K levels, mineral fertilizers were proportionately reduced, while emissions from FYM application (machinery use, direct emissions to air and water) were included.

Table 5-1: Scenarios description

Scenario	Description
BAU (Business-As-Usual)	Conventional Po Valley system with heavy tillage, mineral fertilizers, and no cover crops. Crop residues are the sole carbon input, serving as the baseline for comparison.
CC (Cover Crops)	Introduction of cover crops between main crops to increase soil carbon input and cover. Involves extra field operations but reduces nutrient losses and irrigation needs while improving soil structure.
RT (Reduced Tillage)	Reduction of soil disturbance to slow organic matter decomposition and enhance SOC retention. Incorporates lower fuel use but possible yield penalties during transition years.
FYM (Farmyard Manure)	Partial replacement of mineral fertilizers with manure to increase organic carbon inputs. Includes emissions from

manure handling and application, with nutrient balance maintained.

5.2.2 RothC model parametrization

RothC-26.3 is a process-based model for simulating organic carbon dynamics in non-waterlogged topsoils. The model simulates changes in five distinct organic carbon pools and total Soil Organic Carbon (SOC). The inputs required include soil texture, carbon inputs, climate data, vegetation cover, and the decomposability of incoming plant material. The five organic carbon pools modeled are decomposable plant material (DPM), resistant plant material (RPM), microbial mass (BIO), humified organic matter (HUM), and inert organic matter (IOM) (D. S. Jenkinson et al., 1987, 1991, 1992, 1997; Jenkinson and Coleman, 1994). A critical parameter in the model is the DPM/RPM ratio, typically set at 1.44 for most agricultural crops, indicating that plant material is composed of 59% DPM and 41% RPM. Decomposition rates for DPM, RPM, BIO, and HUM are influenced by climate, clay content, and vegetation cover. The model operates in two modes: an inverse mode, which calculates the annual carbon input required to achieve a specified equilibrium SOC content, and a forward mode, which simulates SOC dynamics based on predefined monthly carbon inputs.

RothC-26.3 was implemented using R version 4.0.3, specifically within the SoilR package (Carlos et al., 2012; R Core Team, 2023). The default SoilR model was modified to include the effects of soil cover by adjusting the moisture function with a "vegetation cover factor", which was derived from rescaled NDVI values, where 0 represents bare soil and 1 represents fully vegetated soil. A threshold of 0.8 was established to determine full vegetation cover. RothC-26.3 provides a robust framework for simulating soil organic carbon dynamics, incorporating key environmental and biological factors, and its adaptability is enhanced through the modification of the SoilR package to include soil cover effects. The model was first applied in inverse mode over a 10,000-year reference period to estimate the annual carbon inputs needed to achieve initial SOC levels, assuming that SOC was in equilibrium (“spin-up phase”). CF practices were then modelled by parameterising the RothC model.

The first scenario analyses the adoption of reduced tillage practices, such as no-tillage or reduced tillage, which decreases the decomposition rates of crop residues and thereby increases soil carbon stocks. Tillage Rate Modifier (TRM) factors were used to simulate the adoption of no-till and reduced tillage practices. These modifiers adjust the decomposition rate constants that control how carbon moves between soil carbon pools. In the RothC model, Soil Organic

Carbon (SOC) is divided into different pools that represent different levels of decomposition. The decomposition of these pools over time was modeled by applying a set of rate modifiers. The equation used to describe this process is structured to account for several factors influencing decomposition, including temperature, moisture, and soil cover, represented by specific rate modifiers - Equation 5.1.

$$SOC_{t+1} = SOC_t \times e^{-abcdkt} \qquad \text{(Eq. - 5.1)}$$

Where SOC_t represents SOC [t C ha $^{-1}$] at time t and SOC_{t+1}represents SOC [t C ha $^{-1}$] at time t + 1. The k factor is constant and differs according to the specific compartment annual decomposition rate: 0.02 for HUM pool, 0.66 for BIO pool, 0.3 for RPM pool and 10 for DPM pool. The exponents are the rate modifiers: a for temperature, b for moisture, and c for soil cover. All exponents are unitless (Coleman and Jenkinson, 1996). In this equation the factor d is the TRM. Research showed that no-till practices correspond to a TRM of 0.95, while reduced tillage has a TRM of 0.93, relative to conventional high-intensity tillage with a default TRM of 1 (Jordon and Smith, 2022). The Tillage Rate Modifier (TRM) factors are used to model the specific response to tillage practices. For instance, no-till or reduced tillage practices reduce the decomposition rate by reducing soil disturbance, and this effect is incorporated into the TRMs. As a result, the TRMs modify the rate at which carbon is cycled through each SOC compartment, allowing more precise simulation of soil carbon changes as tillage intensity is reduced (Hyun and Yoo, 2024). Empirical data on the impacts on yields of conservative tillage systems was included to capture agronomic trade-offs. According to previous studies (Pittelkow et al., 2015) the average yield reduction across 50 crops was 5.1%, with crop-specific reductions of -2.6% for wheat, and -7.6% for maize. These patterns are supported by more recent analyses, which show reductions of between 6% and 20% during the initial transition years (Van Balen et al., 2023; Yue et al., 2023). However, as soils adapt and crop rotations improve, these effects, which depend on the environmental context, may diminish over time.

The main factor contributing to the change in soil organic carbon (SOC) following the adoption of cover crops is the increased carbon input they provide, while other minor effects are typically not considered (Bolinder et al., 2020). To account for this, the carbon inputs have been adjusted to reflect the additional organic matter from the cultivation of cover crops. However, there is limited research specifically addressing the potential of cover crops to enhance carbon inputs under the conditions of northern Italy in the Po Valley. An analysis from western Germany found a 12% increase in carbon inputs from cover crops compared to the business-as-usual (BAU) scenario (Seitz et al., 2023). Due to the lack of specific data for the study area, a conservative value of 10% increase in carbon inputs was assumed for the present analysis. The DPM/RPM ratio, which influences the rate of decomposition of plant material, is set to the default value of 1.44 for agricultural land, according to Coleman and Jenkinson (1996) and other studies (Jebari et al., 2021; Fantin et al., 2022).

Organic fertilization was modelled using regional data from the Po Valley (Italy). The application rate was assumed to be 40 t ha^{-1} of Farmyard Manure (FYM), characterized by an average moisture content of 93.3% and a carbon content on a dry basis of 33.1% (Triberti et al., 2016). This resulted in a carbon input from FYM of 0.89 tC ha^{-1}. The contribution of FYM was intended as an additional carbon input which is assumed to be more decomposed than normal crop plant material. In the model, FYM was split into DPM 49%, RPM 49% and HUM 2%, according to Coleman and Jenkinson (1996).

5.2.3 RothC Input Data

The model requires data on climate (air temperature, precipitation, and potential evapotranspiration) and soil (clay content and initial SOC content). Climate data, including temperature and precipitation, were obtained from the NASA Power Service (Hegyi et al., 2024). Potential evapotranspiration was calculated using the Penman-Monteith equation (FAO, 1998). Soil data, such as the initial SOC content in the topsoil (0-30 cm) and clay, were obtained from the International Soil Reference and Information Centre (ISRIC) SoilGrids 2.0, with a spatial resolution of 250 m (Poggio et al., 2021). Clay data are available at three different depths (0-5 cm, 5-15 cm and 15-30 cm) and were integrated over soil depths up to 30 cm using the trapezoidal rule (Bilas et al., 2022). RothC requires monthly Vegetation Cover (VC) information. VC data were obtained from the MOD13A2 Version 6 product, which provides the Normalized Difference Vegetation Index (NDVI) at a spatial resolution of 1 km (Didan, 2015). Google Earth Engine was used to retrieve satellite data and calculate the monthly probability of ground vegetation cover (NDVI > 0.6).

5.2.4 Life Cycle Assessment

Life Cycle Assessment (LCA) is a scientific approach to decision making that quantifies the potential environmental impacts of products and services over their entire life cycle. The evaluation includes all stages of the product's life cycle, including raw material production, manufacturing, use, and end-of-life (Del Borghi, 2013). The steps involved in an LCA are the definition of the objective and scope, the inventory analysis, the impact assessment, and the interpretation of the results (ISO, 2006a,b).

For this LCA, the functional unit is defined as 1 hectare of crop cultivation using conventional agriculture or CF practices. A cradle-to-farmgate methodology was chosen. **Figure 5.1** displays the processes that form part of the system boundaries. The supply and conversion of fuels, energy, and raw materials into finished agricultural equipment, infrastructure, and crop inputs-such as seeds, fertilizers, and tractors-are the background processes (Del Borghi et al., 2020). This study covers the following agricultural practices: tillage, sowing, crop protection, fertilizing, and harvesting.

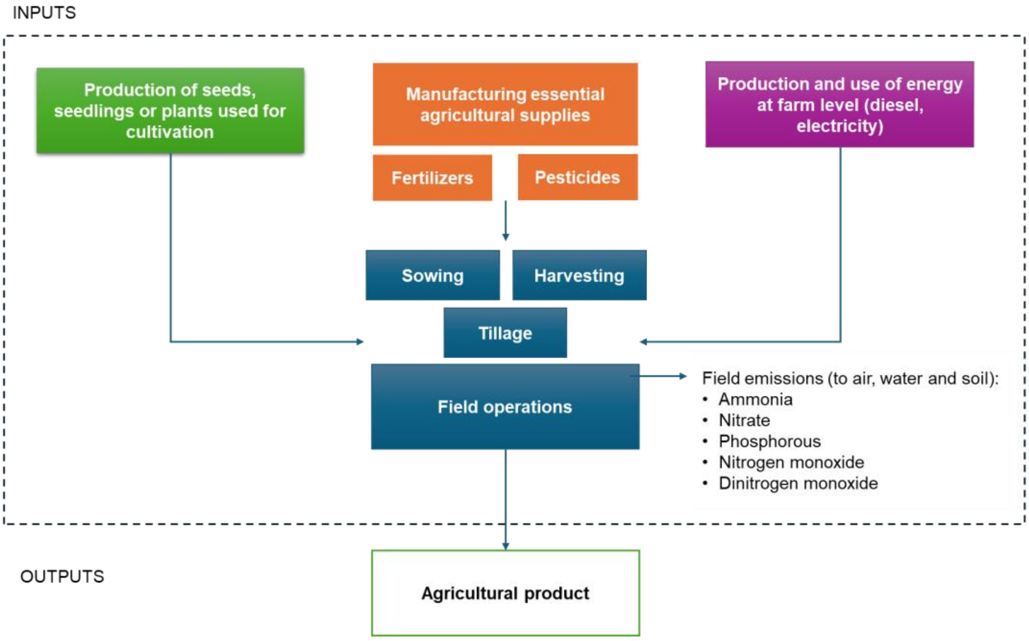

Figure 5-1: Life Cycle Assessment system boundaries

Life Cycle Assessment (LCA) was conducted using SimaPro 10.2 software, with the Ecoinvent 3.11 database (Frischknecht and Rebitzer, 2005; Wernet et al., 2016) as the source of secondary life cycle inventory data. Environmental impacts were assessed using the Environmental Footprint 3.1 impact assessment methodology (JRC, 2023), and a subset of relevant impact categories was selected according to their relevance to agricultural systems.

The collection and processing of agricultural data for the LCA calculation focuses on the inventory data of the cultivation phase for the Business-as-Usual (BAU) scenario. The BAU scenario, which is the baseline, differs for each field but in most cases includes conventional tillage operations, no use of organic fertilizer, and no use of cover crops. This provided a baseline against which improvements and interventions in different scenarios could be assessed. The main data collected related to water, fertilizers, chemical treatments and diesel use. Secondary data, however, were used to model the quantity of seeds and seedlings. Primary data were collected directly from the HelpSoil project, while secondary data were collected from private industry reports and from the Ecoinvent database version 3.11. These are critical to developing a complete life cycle inventory and ensuring the accuracy of the data. The emissions due to fertilizer application were calculated according to the rules set out in the 2019 IPCC Guidelines, Volume 4: Agriculture, Forestry and Other Land Use (AFOLU), Chapter 11 (IPCC, 2019). Nutrient emissions were considered in terms of nitrate, phosphate, and gaseous emissions such as nitrous oxide from nitrogen fertilizer application. Both direct and indirect N_2O emissions were accounted for. Indirect emissions included N_2O emissions from the atmospheric deposition of N on soils and water surfaces, and emissions from N leaching and runoff. In most of the fields included in the study, all of the fertilizers used in the BAU scenario were inorganic, consisting of nitrogen-, phosphatic- and potassium-based fertilizers. For each of the CF scenarios studied, the inventory data were modified, and assumptions were made to calculate the associated project emissions. The analysis considered the four scenarios presented in chapter 2.1: BAU, RT, CC and FYM, for which the detailed Life Cycle Inventory (LCI) is reported in **Table 5.2.**

In the CC scenario, emissions were primarily associated with additional field operations, including sowing, mowing, and biomass incorporation. These operations increase diesel consumption by approximately 30 liters per hectare (after consultation with experts), resulting in indirect GHG emissions from diesel production and direct emissions during equipment use. While indirect GHG emissions from seed production were considered negligible, cover crops provide benefits such as improved soil organic matter, structure, and nutrient availability. The adoption of cover crops allows also for reductions in nitrate leaching in the order of 50-70% (Nouri et al., 2022; Tonitto et al., 2006). To reflect this, the factor for nutrient losses reduction implied was 60%. Other benefits of the use of cover crops are about improved water use, as cover crops can reduce water requirements. In fact, literature on irrigated systems shows that the use of cover crops can decrease the annual irrigation demand by 10-25% compared to bare-fallow fields (Gabriel et al., 2012; Novara et al., 2021). Based on this evidence, in the CC scenario irrigation requirements are lowered by 15%. Other input data, with the exception of diesel consumption, leaching of nutrients, water use and pesticide application, were assumed to be unchanged from the BAU scenario.

Table 5-2: Life Cycle Inventory for the four scenarios analyzed.

INPUT	**U.M.**	**BAU Scenario**	**CC Scenario**	**RT Scenario**	**FYM Scenario**
Seeds/Seedling	kg/ha	372.11	522.11	372.11	372.11
Diesel	l/ha	71.51	92.96	35.75	71.51
Water	m^3/ha	1509.67	1283.22	1509.67	1509.67
Fertilizer: UREA	kg/ha	79.25	79.25	79.25	30.70
Fertilizer: Ammonium Nitrate	kg/ha	16.52	16.52	16.52	5.90
Fertilizer: nitrogen based	kg/ha	17.66	17.66	17.66	6.70
Fertilizer: Phosphoric Anhydride	kg/ha	20.16	20.16	20.16	8.01
Fertilizer: Diammonium phosphate	kg/ha	8.23	8.23	8.23	3.29
Fertilizer: Ammonium sulphate	kg/ha	2.08	2.08	2.08	0.83
Fertilizer: Potassium oxide	kg/ha	12.91	12.91	12.91	5.13
Fertilizer: Potassium chloride	kg/ha	11.25	11.25	11.25	4.50
Fertilizer: Organic	kg/ha	4573.86	4573.86	4573.86	40000.00
Pesticides	kg/ha	9.95	19.90	9.95	9.95
Nutrient/toxicity emissions					
NH_3 to air	kg/ha	20.80	20.80	20.80	56.41
NO to air	kg/ha	24.63	24.63	24.63	117.87
NO_3^- to water	kg/ha	98.78	59.27	98.78	589.17
P to water	kg/ha	1.52	1.52	1.52	0.61
Pesticides	kg/ha	9.95	19.90	9.95	9.95
GHG emissions					
Direct N_2O from soils	kg/ha	3.62	3.62	3.62	6.49
N_2O from N deposition	kg/ha	0.29	0.29	0.29	1.01
N_2O from NO_3^- leaching/runoff	kg/ha	0.17	0.10	0.17	1.00

In the RT scenario, project emissions account for reduced diesel consumption due to fewer tractor passes in the field. Based on literature values, reduced tillage can decrease fuel use by approximately 50% compared to conventional practices (Lal, 2004; Keshavarz Afshar and Dekamin, 2022). Therefore, a 50% reduction in diesel consumption has been applied in this scenario. All other input data, except diesel consumption, were assumed to be unchanged from the BAU scenario (**Table 5.2**).

In the FYM scenario, emissions considered include those associated with the use of agricultural machinery to apply organic fertilizer, as well as emissions from the fertilizers. These include indirect greenhouse gas (GHG) emissions of organic fertilizers (no upstream impact allocated, including only transport to the farm) and direct emissions to air and water after application. As the substitution of inorganic fertilizers is partial (ensuring appropriate N-P-K levels), the emissions from fertilizer application have been treated as substitution rather than additional emissions compared to the BAU scenario, with an unchanged total amount of fertilizer. Emissions to air and water from the use of organic fertilizers were calculated according to IPCC guidelines. For direct emissions calculation an average 0.6% nitrogen content was assumed.

5.2.5 Integration of model simulations and Life Cycle Assessment

The RothC model simulates SOC dynamics. Each scenario in this study provides an estimate of the annual carbon sequestered in the soil or lost to the atmosphere (referred to as the annual delta SOC change and expressed in t C ha-1 yr-1). Therefore, the average of the annual SOC changes over the study period was used to calculate the mean delta SOC change. To integrate the soil carbon dynamics results with the LCA results, the Carbon Footprint approach is used (Cammarata et al., 2025), which allows the calculation of net GHG emissions expressed as t CO_2 -eq $ha^{-1}yr^{-1}$. The atomic masses of carbon (12 g mol-1) and carbon dioxide (44 g mol^{-1}) were used to convert the mean delta SOC change to t CO_2-eq ha^{-1} yr^{-1}, along with the Global Warming Potential - 100 years (GWP) of carbon dioxide per unit equal to one, with the following equation:

$$CO_{2-eq}[ha^{-1}yr^{-1}] = C \times \frac{44}{12} \qquad \text{(Eq. - 5.2)}$$

Where C is the mean annual SOC stock change simulated by RothC-26.3 (t C ha-1 yr-1), i.e., the average of the annual delta SOC changes over the study period, which represent the only mechanism through which soils can act as a long-term sink for atmospheric CO_2. In parallel, the LCA framework comprehensively accounts for all greenhouse gas (GHG) emissions associated with agricultural inputs and field processes, including CO_2, CH_4, and N_2O.

To obtain the net carbon balance (CB) (Eq. 5.3), SOC stock changes from RothC were expressed in CO_2 equivalents and compared directly to the total GHG emissions from the LCA. This integration is methodologically consistent, as all fluxes are converted to the same metric (CO_2 eq), and reflects the fact that soil sequestration is the only biophysical mitigation service, whereas the farming system generates multiple GHG emissions. The purpose of this combined approach is not to homogenize emission sources, but rather to evaluate whether the SOC sequestration potential can offset the GHG emissions of the system over its entire life-cycle. The CB was then obtained by adding the carbon input (from the soil carbon models) and the carbon output (from the LCA):

$$CB[CO_2 ha^{-1} yr^{-1}] = C_{Ei} - C_{s_i} \qquad \text{(Eq. - 5.3)}$$

Where C_E is the CO_2-equivalents emitted to the atmosphere by the i-scenario and C_{Si} is the CO_2-equivalents that are either sequestered in the soil (i.e. negative impact on CB) or lost to the atmosphere (i.e. positive impact on CB). In this study, CE denotes the total life-cycle climate change impact (total GHG emissions) returned by the LCA model. It aggregates direct on-field and upstream supply-chain emissions for inputs and operations and is expressed in t CO_2-eq ha-1 yr-1. CSi is taken as a positive value when C is sequestered and as a negative value when the soil is a source of emissions. The dynamic (i.e., C sequestered or lost on an annual basis) output of the carbon models was averaged over years to allow combination with the LCA output.

In addition to GHG emissions to the atmosphere, the following environmental impact indicators were assessed: Acidification, Water use, Resource use (fossil), Photochemical ozone formation, Land Use and Eutrophication (terrestrial). These results are not directly linked to the SOC results but should be seen as additional key information. As such, they can provide a broader view of the environmental impacts associated with the implementation of CF practices. This view provides insight into whether these practices actually benefit the environment or simply shift environmental impacts to other environmental compartments.

5.2.6 Statistical and uncertainty analysis

The outputs of the RothC model were subjected to a statistical analysis of the dynamics of soil organic carbon (SOC). Cumulative changes in SOC over 20 years (ΔSOC, $t\ C\cdot ha^{-1}$) were computed at 16 independent field points for each management scenario (BAU, CC, FYM, and RT). Box-and-whisker plots were used to visualize the distributions and compute descriptive statistics, including the mean, standard deviation, standard error, and 95% confidence intervals. A repeated-measures ANOVA was applied to test for differences among management practices, with significance assessed at $\alpha = 0.05$. To assess the impacts of management practices, effect sizes (partial η^2) were reported. To evaluate temporal sequestration dynamics, SOC trajectories were also used to derive linear slope analysis ($t\ C\cdot ha^{-1}\cdot yr^{-1}$) and the area under the curve (AUC, $t\ C\cdot ha^{-1}\cdot 20\ yr^{-1}$).

SimaPro 10.2 was used to perform uncertainty analysis for the life cycle assessment (LCA) at a 95% confidence level using a Monte Carlo simulation ($n = 2,500$ runs). Through the inventory and impact assessment, this method spreads model uncertainty and parameter variability, yielding probability distributions for the environmental impacts evaluated. Finally, the standard deviation (SD) and the coefficient of variation (CV) were computed. Both statistical variability and model uncertainty were specifically addressed in the integrated carbon balance by integrating LCA-derived emission estimates with RothC-based SOC sequestration results.

5.3 RESULTS

5.3.1 Soil Organic Carbon

The RothC model outputs the total soil organic carbon (SOC) stock [$t\ C\ ha^{-1}$] in the topsoil (0-30 cm). Across the 20-year simulation period, all scenarios displayed a positive trend in SOC accumulation, though with different magnitudes depending on the management practice (**Figure 5.2**). The Business as Usual (BAU) scenario showed only a slight increase, while all Carbon Farming (CF) practices enhanced SOC storage compared to BAU.

To compare overall sequestration performance among scenarios, the mean ΔSOC after 20 years was calculated (**Figure 5.3**). The BAU scenario resulted in a mean increase of 0.55 $t\ C\ ha^{-1}$ (±0.31 $t\ C\ ha^{-1}$), whereas Cover Crops (CC) led to an increase of 1.73 $t\ C\ ha^{-1}$ (±0.30 $t\ C\ ha^{-1}$). Farmyard Manure (FYM) showed the highest SOC accumulation with 4.89 $t\ C\ ha^{-1}$ (±0.24 $t\ C\ ha^{-1}$), while

Reduced Tillage (RT) resulted in a mean increase of 1.34 t C ha⁻¹ (±0.32 t C ha⁻¹).

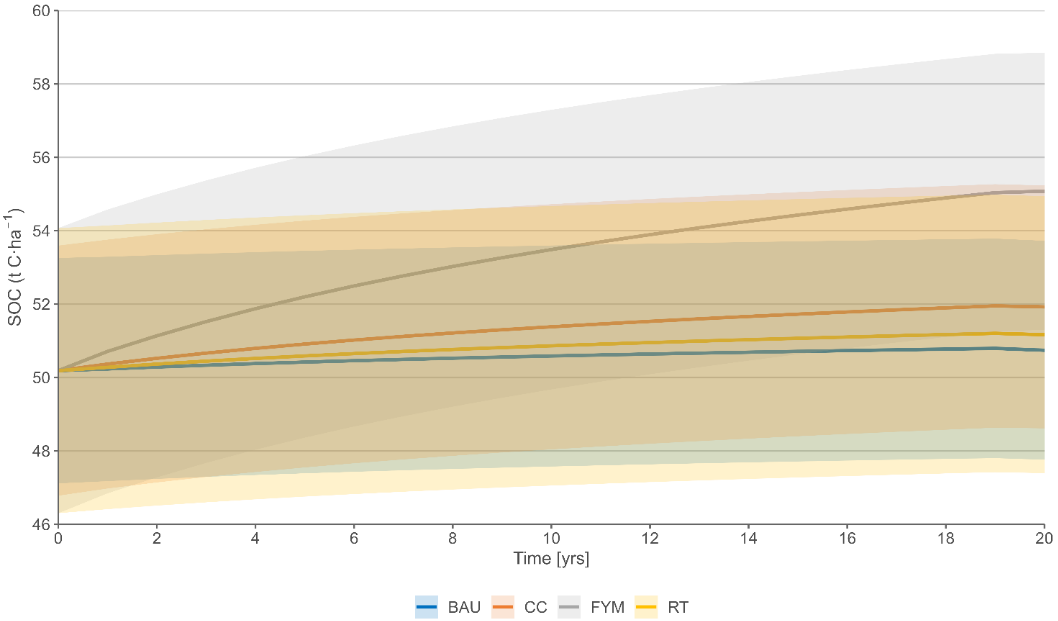

Figure 5-2: Simulated soil organic carbon (SOC) stocks (t C ha⁻¹) in the 0-30 cm soil layer over a 20-year period under different management scenarios: Business as Usual (BAU), Cover Crops (CC), Farmyard Manure (FYM), and Reduced Tillage (RT). Lines represent mean values across replicates and shaded bands represent the error.

The distribution of SOC changes across replicates for each scenario is represented by the box-and-whisker plots (**Figure 5.3**). The crosses represent the mean values. This graphical representation not only highlights the central tendency of SOC changes, but also illustrates the variability among the simulations, providing a comprehensive overview of the modeled outcomes.

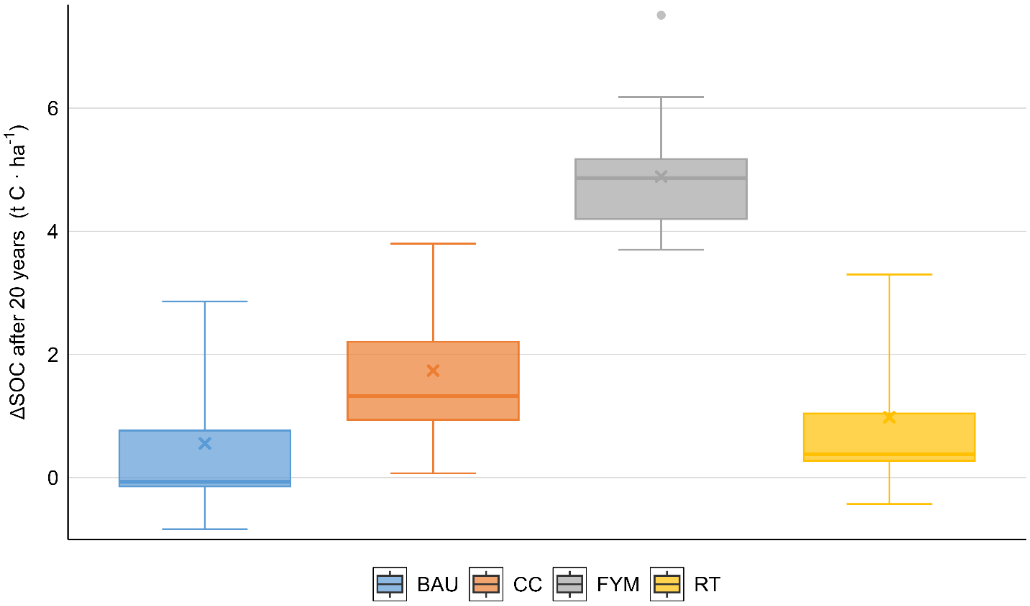

Figure 5-3: Distribution of cumulative SOC changes (ΔSOC, t C ha $^{-1}$) after 20 years for each management scenario. Boxplots show the interquartile range (25th-75th percentile, Q1 - Q3), horizontal lines represent medians, whiskers indicate non-outlier range, points are outliers; mean markers are shown. n = 16 per scenario.

The descriptive statistics and ANOVA of the cumulative change in soil organic carbon (ΔSOC) after 20 years for the four management scenarios (n = 16 independent field points per scenario) are reported in **Table 5.3**. A repeated-measures ANOVA showed a highly significant effect of management on cumulative ΔSOC ($p < 1\times10^{-18}$), with partial $\eta^2 = 0.86$ indicating a very large effect size. The FYM treatment shows a distinct and significantly higher SOC accumulation compared to the other scenarios, while Cover Crops (CC) and Reduced Tillage (RT) present intermediate performances between FYM and BAU.

Table 5-3: Descriptive statistics and ANOVA for the RothC simulation outcomes

Scenario	**BAU**	**CC**	**FYM**	**RT**
Descriptive Statistics				
Mean ΔSOC [t C ha⁻¹]	0,56	1,74	4,89	0,98
Standard Deviation [t C ha⁻¹]	1,25	1,19	0,97	1,30
Standard Error [t C ha⁻¹]	0,31	0,30	0,24	0,32
C.I. 95% [t C ha⁻¹]	0,67	0,63	0,52	0,69
ANOVA				
p-value		$5,76 \times 10^{-19}$		
Partial η^2		0,86		

Table 5-4: Slope and AUC values for the four investigated scenarios

Scenario	**Slope** [t C·ha⁻¹·yr⁻¹]	**AUC** [t C·ha⁻¹·20 yr⁻¹]
BAU	0,03	7,37
CC	0,09	22,06
RT	0,05	12,50
FYM	0,24	60,99

The results of the slope analysis indicate the annual rate of soil organic carbon (SOC) sequestration under different agricultural practices (**Table 5.4**), with FYM demonstrating the highest average annual increase (0.24 t C·ha⁻¹·yr⁻¹), significantly outperforming conventional agriculture (BAU, 0.03 t C·ha⁻¹·yr⁻¹). The AUC values, which represent the cumulative SOC sequestration over 20 years, further confirm these findings, with FYM accumulating nearly nine times more carbon than BAU. Cover crops (CC) and reduced tillage (RT) provide moderate improvements but remain substantially lower than FYM.

5.3.2 Life Cycle Assessment

The full set of environmental impact indicators from the LCA analysis is reported in **Table 5.5**, presenting average values, standard deviations (SD), and coefficient of variation (CV) from the uncertainty analysis across all scenarios. **Table 5.5** includes eleven environmental impact categories that capture different aspects of agricultural system performance beyond climate change. These categories were selected to provide a comprehensive view of potential environmental burdens across multiple emission pathways and environmental compartments: atmospheric impacts (acidification, photochemical ozone formation), aquatic impacts (eutrophication in marine and freshwater environments), terrestrial impacts (terrestrial eutrophication), resource depletion (fossil fuel consumption), and water consumption.

The LCA results (**Table 5.5**) demonstrate substantial variation in environmental impact magnitudes across the four management scenarios. Climate change impacts ranged from 2.12 t CO_2 eq ha^{-1} yr^{-1} for RT to 3.25 t CO_2 eq ha^{-1} yr^{-1} for FYM, with BAU and CC showing intermediate values (2.30 and 2.31 t CO_2 eq ha^{-1} yr^{-1}, respectively). However, the magnitude of differences varies dramatically across impact categories. For acidification, impacts ranged from 41.13 mol H^+ eq (RT) to 150.53 mol H^+ eq (FYM), representing approximately 3.7-fold variation. Marine eutrophication showed even greater variation, ranging from 40.26 kg N eq (CC) to 215.25 kg N eq (FYM), a 530% difference.

The breakdown of the main GHG emission sources is shown in **Figure 5.4**. In all scenarios, direct emissions from fertilizers are the main source of carbon emissions, accounting for more than 40% of the total emissions and reaching 71% in the FYM scenario. The other relevant sources of carbon emissions are in descending order: fertilizer production, diesel consumption for field operations, pesticides, and seeds or seedlings.

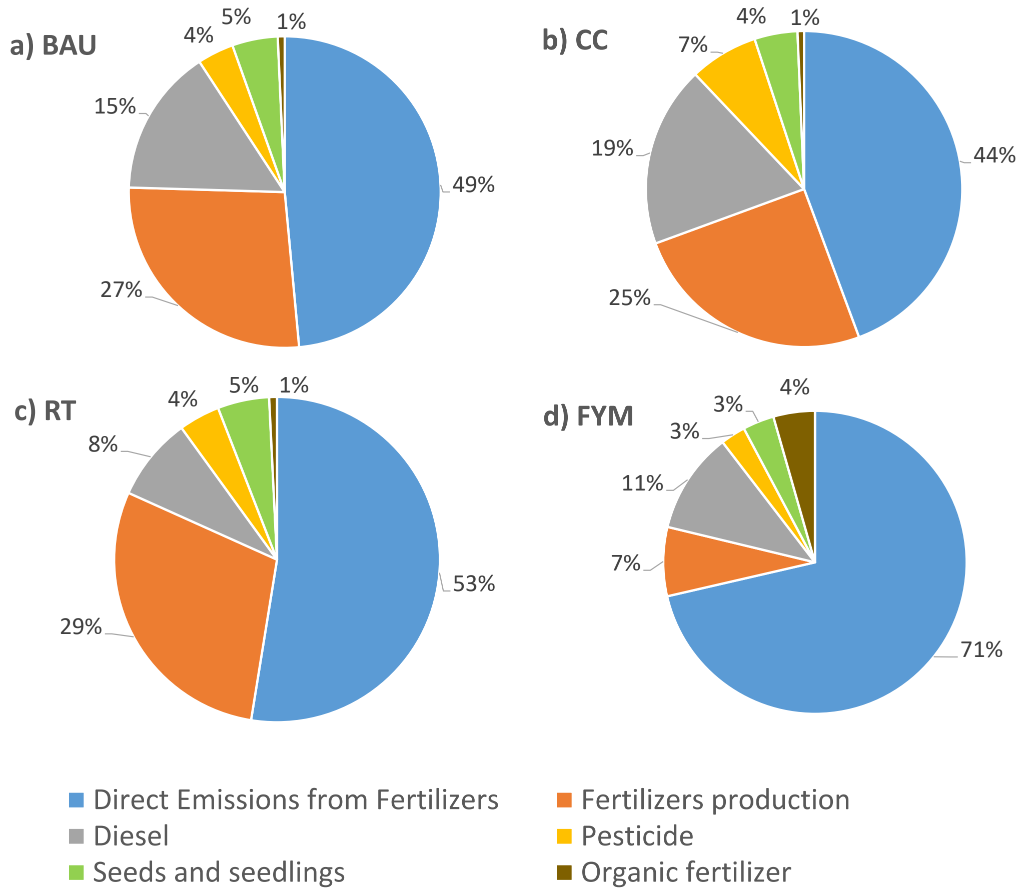

Figure 5-4: Climate change impact: contribution of emission sources for four land management scenarios (BAU, CC, FYM, RT), expressed in t CO_2-eq ha^{-1} yr^{-1}. Each segment represents the share of total GHG emissions from each source category. a) BAU, Business As Usual; b) CC, cover Crops; c) RT, Reduced Tillage; d) FYM, FarmYard Manure.

Table 5-5: Life Cycle Impacts for the four analyzed scenarios (FU: 1ha), table reports the average, SD and CV coming from the uncertainty analysis.

Impact Category	U.M.	BAU		
		Average	**SD**	**CV**
Acidification	mol H^+ eq	42.51	1.52	3.58%
Climate change	t CO_2 eq	2.30	0.19	8.19%
Eutrophication, marine	kg N eq	45.58	0.12	0.27%
Eutrophication, freshwater	kg P eq	1.90	1.29	67.73%
Eutrophication, terrestrial	mol N eq	371.18	5.23	1.41%
Land use	Pt	61322.53	129000.00	210.36%
Ozone depletion	kg CFC_{11} eq	2.01E-04	7.77E-05	38.72%
Particulate matter	disease inc.	5.81E-04	2.13E-05	3.66%
Photochemical ozone formation	kg NMVOC eq	32.11	1.34	4.17%
Resource use, fossils	MJ	18883.64	1650.00	8.74%
Water use	m^3 depriv.	69319.67	13400.00	19.33%

Impact Category	U.M.	CC		
		Average	**SD**	**CV**
Acidification	mol H^+ eq	47.01	1.73	3.68%
Climate change	t CO_2 eq	2.31	0.21	9.08%
Eutrophication, marine	kg N eq	40.26	0.17	0.41%
Eutrophication, freshwater	kg P eq	2.00	1.65	82.44%
Eutrophication, terrestrial	mol N eq	387.45	7.58	1.96%
Land use	Pt	65378.23	128000.00	195.78%
Ozone depletion	kg CFC_{11} eq	3.71E-04	1.70E-04	45.83%
Particulate matter	disease inc.	6.11E-04	2.03E-05	3.32%
Photochemical ozone formation	kg NMVOC eq	33.92	1.61	4.75%
Resource use, fossils	MJ	22228.64	2190.00	9.85%
Water use	m^3 depriv.	59162.58	14200.00	24.00%

Impact Category	U.M.	RT		
		Average	SD	CV
Acidification	mol H^+ eq	41.13	1.12	2.72%
Climate change	t CO_2 eq	2.12	0.17	7.93%
Eutrophication, marine	kg N eq	45.04	0.12	0.27%
Eutrophication, freshwater	kg P eq	1.88	1.10	58.53%
Eutrophication, terrestrial	mol N eq	365.32	3.59	0.98%
Land use	Pt	58802.07	116000.00	197.27%
Ozone depletion	kg CFC_{11} eq	1.98E-04	9.28E-05	46.78%
Particulate matter	disease inc.	5.73E-04	1.68E-05	2.93%
Photochemical ozone formation	kg NMVOC eq	30.33	0.75	2.47%
Resource use, fossils	MJ	16614.00	1860.00	11.20%
Water use	m^3 depriv.	69305.65	11200.00	16.16%

Impact Category	U.M.	FYM		
		Average	SD	CV
Acidification	mol H^+ eq	150.53	1.08	0.72%
Climate change	t CO_2 eq	3.25	0.15	4.58%
Eutrophication, marine	kg N eq	215.25	0.11	0.05%
Eutrophication, freshwater	kg P eq	0.95	1.18	124.58%
Eutrophication, terrestrial	mol N eq	1274.34	4.99	0.39%
Land use	Pt	60927.37	122000.00	200.24%
Ozone depletion	kg CFC_{11} eq	1.93E-04	8.71E-05	45.23%
Particulate matter	disease inc.	1.46E-03	1.51E-05	1.03%
Photochemical ozone formation	kg NMVOC eq	125.22	1.21	0.97%
Resource use, fossils	MJ	15025.70	1540.00	10.25%
Water use	m^3 depriv.	68905.67	10600.00	15.38%

Reduced tillage emerges as the most robust strategy, demonstrating consistent environmental benefits across acidification, eutrophication, and photochemical ozone formation categories (all reduced relative to BAU) while achieving -20.4% lower fossil fuel consumption (16,614 MJ ha^{-1} yr^{-1}) and maintaining the lowest model uncertainty (CV 0.98-11.20%). Farmyard manure maximizes environmental burdens, with dramatic increases across acidification (+254%), marine eutrophication (+372%), terrestrial eutrophication (+243%), and photochemical ozone formation (+290%), despite achieving the lowest fossil fuel demand (15,025.7 MJ ha^{-1} yr^{-1}). Cover crops present a polarized profile: delivering marine eutrophication and water use improvements (-11.7% and -14.7%, respectively) while incurring acidification and terrestrial eutrophication penalties (+10.6% and +4.4%), coupled with the highest fossil resource use intensity (22,228.64 MJ ha^{-1} yr^{-1}).

5.3.3 Carbon Balance

The carbon balance (CB) was calculated as the algebraic sum of carbon sequestration and system-wide GHG emissions, expressed on an annual basis over the 20-year modeling period. Soil organic carbon gains were converted to CO_2 equivalents using Equation 2. All scenarios resulted in positive SOC sequestration, quantified in **Table 5.6**. BAU showed the lowest sequestration rate at 0.10 t CO_2 ha^{-1} yr^{-1} (±0.06), while CC and RT achieved moderate sequestration of 0.32 t CO_2 ha^{-1} yr^{-1} (±0.05) and 0.25 t CO_2 ha^{-1} yr^{-1} (±0.06), respectively. FYM achieved the highest sequestration rate at 0.90 t CO_2 ha^{-1} yr^{-1} (±0.04), representing a 9-fold increase over BAU.

Table 5-6: Carbon balance components for each scenario, sequestration potential, emissions, and net carbon balance

SCENARIO	CARBON SEQUESTRATION [t CO_2 ha^{-1} yr^{-1}]	CARBON EMISSIONS [t CO_2 ha^{-1} yr^{-1}]	CARBON BALANCE [t CO_2 ha^{-1} yr^{-1}]
BAU	0.10 (±0.06)	2.30 (±0.19)	2.08 (±0.20)
CC	0.32 (±0.05)	2.31 (±0.21)	1.91 (±0.22)
RT	0.25 (±0.06)	2.12 (±0.17)	1.87 (±0.18)
FYM	0.90 (±0.04)	3.25 (±0.15)	1.93 (±0.16)

Despite substantial differences in SOC sequestration rates (ranging from 0.10 to 0.90 t CO_2 ha^{-1} yr^{-1}), the net carbon balance remained relatively similar across scenarios (1.87-2.08 t CO_2 ha^{-1} yr^{-1}). This reflects a trade-off mechanism: while management practices such as FYM and CC enhanced soil carbon sequestration, they were partially offset by increased system-wide emissions. The FYM scenario shows highest sequestration but also highest total emissions at 3.25 t CO_2 ha^{-1} yr^{-1}. Consequently, the ranking of scenarios by sequestration potential did not translate into proportional improvements in overall carbon balance.

5.4 DISCUSSION

5.4.1 Soil Organic Carbon Sequestration Potential

Several agricultural management practices were assessed in previous research in relation to their sequestration potential, including agroforestry, hedgerow planting, cover crops, and reduced tillage, and demonstrated increases in SOC in several scenarios (Aertsens et al., 2013). The results of this study confirm that different land management regimes have significant impacts on soil carbon sequestration and GHG fluxes. The results of the RothC model reveal a general SOC increase across all scenarios, with the farmyard manure (FYM) scenario sequestering, on average, the most carbon (4.89 t C ha^{-1}) after 20 years. This is consistent with evidence indicating that organic amendments such as manure enhance soil carbon sequestration (Owusu et al., 2024). CC and RT also increase SOC, but to a lesser extent, confirming their value as sustainable options (Babu et al., 2023; Breil et al., 2023). The BAU scenario shows a moderate increase in SOC stocks. Although conventional agriculture is often associated with SOC losses (Francaviglia et al., 2018), in the BAU scenario the arable crop rotation included both maize and wheat which are known for adding large amounts of crop residues contributing to SOC (J. Wang et al., 2015) and some exogenous FYM was also included in the BAU scenario, since three of the farms were already using FYM as part of their fertilization strategy.

RothC-26.3 was configured for Northern Italian arable systems with a 20-year horizon and 0-30 cm topsoil, chosen to match policy-relevant evaluation windows (Ledo et al., 2020; Petersson et al., 2025) and to represent the management-responsive layer in Mediterranean croplands. RT, CC, and FYM were selected as complementary sequestration pathways and parameterized consistently with recent European RothC adaptations (Pesce et al., 2024). These adaptations align SOC modeling with actionable agronomic levers in Northern Italy while extending prior syntheses that identify input-driven and disturbance-mediated pathways as primary routes to SOC gains in croplands. The carbon sequestration values in the topsoil obtained from the simulations performed in this study are in line with previous findings, which are in the same order of magnitude for FYM, CC and RT practices (Bolinder et al., 2020). In addition, nitrogen inputs from crop residues have the potential to control SOC stabilization in the long term (Van De Vreken et al., 2016).

While this study successfully modeled SOC dynamics in the topsoil (0-30 cm) with regional relevance, future research should explore the role of lateral carbon fluxes in modeling SOC stocks. This can have an influence on soil erosion and affect sequestration estimates. If the study area is prone to erosion, including these variables could provide a more comprehensive view of SOC dynamics by taking into account both vertical and horizontal carbon fluxes in order to accurately estimate sequestration potential, further confirming the need for integrated assessments in soil carbon science (Nadeu et al., 2015). Restricting analysis to topsoil may underestimate SOC change, underscoring the importance of deeper carbon storage for long term sequestration. Tillage depth is another important variable to consider. The results of this study relate to the first 30 cm of soil, but existing literature emphasizes the importance of including deeper soil horizons to fully quantify SOC stock changes (Ottoy et al., 2016). Future work could extend modeling to 0-60 or 0-100 cm using modified RothC parameterizations or comparable depth-differentiated approaches which would capture mineral-associated carbon stabilization in deeper horizons (Smith et al., 2010). Another important consideration is the potential influence of climate change on SOC dynamics. Conditions of turnover shifts could negatively affect efforts to increase carbon sequestration (Pohanková et al., 2022). As their study suggests, it will be essential to include future climate conditions, as these will affect long-term SOC stability and make SOC accumulation vulnerable to climate-induced shifts, confirming the relevance of climate adaptation strategies.

5.4.2 Environmental Trade-Offs

he LCA results show that, under the modelling hypotheses of this study, the RT scenario has the lowest overall environmental impact, although it provides lower carbon sequestration than the other CF practices. In contrast, FYM achieves high carbon sequestration but also results in the highest GHG emissions, mainly due to the large quantity of fertilizers used. The prevalence of emissions from nitrogen fertilization-whether from mineral or organic sources-highlights the need to optimize fertilization strategies to minimize environmental trade-offs. Nitrogen losses through leaching and runoff pathways are not unique to manure but represent a broader challenge across all fertilization strategies; however, manure-based approaches typically exhibit higher nutrient loss rates due to elevated total nitrogen content and less controlled release patterns relative to granulated mineral fertilizers. The comprehensive environmental impact assessment reveals that FYM has significantly higher eutrophication (+372%) and acidification (+254%) potentials relative to BAU. These findings reveal that although FYM improves carbon sequestration in soils, it may have poorer overall environmental performance unless practices to reduce nutrient losses via runoff and emissions are applied. An important consideration regarding system boundaries emerges when contextualizing the FYM scenario within broader agricultural systems: the manure applied originates from livestock production, which itself generates substantial greenhouse gas emissions (enteric fermentation, manure management). This study follows an attributional LCA approach with a field-level system boundary, treating manure as a co-product with its burdens already allocated elsewhere in the supply chain. However, consequential LCA frameworks that extend the system boundary to include livestock production would potentially further increase emissions in the FYM scenario (Battini et al., 2016). This distinction is critical for policymakers evaluating CF schemes: the carbon sequestration benefits of FYM must be weighed against the full lifecycle emissions of the livestock system generating the manure, particularly in regions with intensive animal production (Singaravadivelan et al., 2023).

Given the critical importance of nitrate vulnerable zones in Northern Italy, the environmental trade-offs of FYM become especially pronounced. While the 254% increase in acidification and 372% increase in marine eutrophication represent substantial burdens, these impacts are particularly problematic in regions where groundwater is already vulnerable to nitrate contamination. In our study, we assumed an application rate of 40 t ha^{-1} of FarmYard Manure as a strategy to increase carbon inputs into the soil and promote carbon sequestration. The authors recognize that this is a high application rate and acknowledge the complex relationship between carbon sequestration and nitrogen management. This issue becomes even more important in nitrate-vulnerable zones (Quilez et al., 2025). Manure-based fertilization contributes significantly to nitrate leaching to groundwater, with up to 20% of groundwater sources exceeding safe nitrate thresholds. In addition, soil nitrogen saturation can lead to a number of negative effects, including soil acidification, nutrient imbalances, and increased nitrous oxide emissions. This trade-off underscores that while manure amendments can be beneficial for carbon sequestration, they can also increase nitrogen pollution, impacting both local water systems and the atmosphere. Consequently, the application of animal manure in nitrate-vulnerable areas must be carefully managed to optimize carbon sequestration while mitigating nitrogen-related environmental risks (Kros et al., 2024). CC and RT, on the other hand, have more evenly distributed effects, with CC providing benefits in terms of land-use efficiency, while RT shows less dependence on fossil resources due to reduced machinery use.

5.4.3 System Boundaries and Lifecycle Considerations

The LCA results indicate that while CF practices can build SOC, they also lead to higher emissions in many environmental impact categories. For these reasons, a comprehensive assessment of agricultural management is needed, as it is not sufficient to focus solely on carbon sequestration potential; potential negative side effects, such as increased GHG emissions from fertilization, must also be assessed. The results of this study are related to burden shifting considerations, where organic amendments such as manure have the potential to trigger increased nutrient leaching and emissions. The carbon balance calculation confirms that although all the scenarios lead to SOC gains, the net carbon balance is still positive because emissions have higher values than sequestration. This confirms the need for complementary mitigation measures, such as the use of carbon offset projects or increased soil amendment effectiveness, to achieve carbon neutrality.

The carbon balance (CB) can be interpreted as a single indicator offering an easy-to-read result for supporting decision makers and can be potentially integrated in more comprehensive evaluations, such as the Water-Energy-Food-Climate (WEFC) nexus (Moreschi et al., 2024). Obtained CB values are positive in all scenarios, i.e. the total emissions are higher than sequestration, but with different values between the scenarios. The maximum carbon sequestration value of 0.90 t CO_2 ha^{-1} yr^{-1} is obtained from the FYM scenario, which is also characterized by the highest Carbon Footprint (2.38 t CO_2 ha^{-1} yr^{-1}) and an overall better performance compared to the BAU scenario. The CC and RT scenarios also result in a lower CB than the BAU scenario, demonstrating their effectiveness for climate mitigation purposes. The CB concept, intended as a balance between carbon sequestration and carbon emissions, was introduced to assess the performance of regenerative agricultural practices in citrus production in Sicily (Italy) (Cammarata et al., 2025). While that context is not directly comparable to our arable system, those authors reported a more favorable CB under regenerative management. In our scenarios, CF practices increased SOC sequestration but the net CB remained similar across scenarios because higher sequestration was offset by higher emissions. Another example of coupling carbon sequestration modelling (using the RothC model) with LCA showed that SOC sequestration can offset a significant part of the carbon footprint of the farming activity. For instance, in the case of olive groves, SOC in the soil up to 15 cm depth was the main driver for the large reduction in net GHG emissions, again confirming the proposition that SOC processes need to be included in product environmental footprints (PEF). However, as the RothC study on olive production suggests, the inclusion of SOC within PEF methodologies requires more site-specific information, which may prove challenging to obtain and analyze (Fantin et al., 2022).

5.4.4 Limitations and Implications for Decision Making on Agricultural Sustainability

The limitations of this study are primarily related to modelling choices. The RothC configuration used in this study represents the average SOC dynamics in non waterlogged topsoil. It does not, however, explicitly capture subsoil processes, erosion driven lateral carbon redistribution, or fine scale spatial heterogeneity. Therefore, SOC estimates are more robust at the system level than at the field or plot scale (Z. Wang et al., 2015). The analysis is restricted to the 0-30 cm layer and a 20 year time horizon, which may underestimate total SOC changes in systems where deeper mineral associated carbon contributes substantially. It also does not fully reflect potential long term saturation or legacy effects under changing climate conditions (Wan et al., 2011). In addition, the attributional, field-level LCA omits broader market responses and only partially reflects upstream livestock emissions through the selected allocation rules. This implies that the climate and eutrophication burdens of manure-intensive strategies could be underestimated when viewed from a full supply-chain or consequential perspective. Finally, the model is parameterised for Northern Italian maize-wheat rotations and a specific FYM application rate. Therefore, extrapolation to other pedoclimatic conditions, crop systems, and regulatory contexts (e.g. stricter manure limits) should be interpreted with caution (Guerrieri et al., 2026).

Within this context, the findings of this study have several important implications for agricultural policy and practice aimed at mitigating climate change through CF. Rather than prescribing a single "best practice”, the results point towards context-dependent decision-making that considers trade-offs across multiple environmental dimensions. In regions, where the primary objective is carbon sequestration (e.g. through carbon markets or compliance schemes), cover crops emerge as a balanced approach, delivering moderate sequestration (0.32

t CO_2 ha^{-1} yr^{-1}) while minimising negative environmental impacts. Reduced tillage provides benefits through fossil fuel reduction. In contrast, high-rate manure application should be reserved for regions where local or regional nitrogen saturation is not a concern where nitrogen-vulnerable zone restrictions do not apply and where livestock production systems already exist, thereby avoiding net lifecycle emission increases. The carbon balance metric used in this study offers a pragmatic tool for comparing management scenarios but the results should not be interpreted as indicating complete carbon neutrality or climate benefit. As mentioned previously, CF practices in this study reduce net emissions relative to BAU but do not achieve a zero or negative carbon balance. These findings have direct implications for designing Carbon Farming schemes and certification frameworks in Europe, including those emerging under the EU Carbon Removal Certification Framework (CRCF). If incentives and credits are granted solely on the basis of SOC increases, the application of high rate manure may be encouraged, despite this causing substantial nitrogen pollution and providing only partial net climate benefits, especially when upstream livestock emissions are fully accounted for. For agricultural systems to meaningfully contribute to climate targets, additional mitigation measures will likely be necessary alongside soil carbon building, such as precision nutrient management to reduce losses, renewable energy adoption for farm operations, or dietary shifts reducing livestock dependence.

5.5 CONCLUSIONS AND FUTURE PERSPECTIVES

This study demonstrates that agricultural management strategies can generate measurable soil organic carbon (SOC) gains, but with context-dependent trade-offs. For Carbon Farming (CF), overall environmental performance should be evaluated, as CF may place too much emphasis on carbon sequestration without fully considering other environmental impacts that require mitigation. Decision-makers should therefore adopt a multi-criteria assessment framework rather than optimize a single indicator, in order to provide a more comprehensive assessment of agricultural sustainability. CF should be viewed as one tool within a broader portfolio of agricultural, dietary and energy-system changes necessary to meet climate targets. Future research must address critical limitations related to climate-adaptation dynamics, to ensure SOC stability under future conditions, and improve monitoring, reporting and verification (MRV) technologies, including remote sensing and machine learning, to enhance SOC inventory accuracy and policy enforcement. Effective policy design also requires integration of CF into existing instruments (e.g. the

Common Agricultural Policy and the Carbon Removal Certification Framework), with explicit recognition of the associated environmental trade-offs. Long-term incentives that balance sequestration goals with farmer profitability are essential, as SOC saturation occurs over decades. Optimised nitrogen management-through precision application, manure-management controls and combined practices such as reduced tillage and agroforestry-can prevent environmental degradation while sustaining SOC gains. Integrating Life Cycle Assessment offers a pathway toward comprehensive sustainability assessment but requires resolving methodological issues related to permanence, data availability and social-equity dimensions. Aligning research, technology development and policy timelines will enable agricultural systems to evolve towards sustainability without compromising soil health or long-term productivity.

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CHAPTER 6

Environmental and social sustainability assessment of a circular process for the valorisation of sewage sludge ash and mining by-products into bio-based fertilisers

This chapter is extensively based on the following publication, under review as of February 2026:

Esposito L., Gagliano E., Tacchino V., **Spotorno S.**, Canziani R., El Chami D., Gallo M., Del Borghi A., Turolla A. (2026). *Environmental and social sustainability assessment of a circular process for the valorisation of sewage sludge ash and mining by-products into bio-based fertilisers* (*Journal of Cleaner Production*)

6 ENVIRONMENTAL AND SOCIAL SUSTAINABILITY ASSESSMENT OF A CIRCULAR PROCESS FOR THE VALORISATION OF SEWAGE SLUDGE ASH AND MINING BY-PRODUCTS INTO BIO-BASED FERTILISERS

ABSTRACT

Phosphorus (P) recovery from sewage sludge ash (SSA) represents an interesting solution to P supply concerns. While techno-economic assessments shown promising outcomes, the environmental and social performance of these recovery processes remains insufficiently explored, hindering the market uptake of recovered P in bio-based fertilisers. This work investigates the environmental impacts of producing four commercial granular fertilisers representative of the Italian fertilizer market in 2024 - ENERGEO CV, ENERGEO CV TOP, LITHOZINC and PHEOSCOR - and the social impacts of raw materials supply (e.g., P, Mg, S, Ca, Cl) for fertiliser manufacturing, targeting specific environmental and social impact categories and relevant stakeholders. Impacts were evaluated for a Business-As-Usual (BAU) scenario, in which fertilisers are based on mineral P (i.e., phosphorite), and a Circular Economy scenario (CE), in which phosphorite is partially replaced with P recovered from SSA via wet chemical extraction and co-precipitated with calcium hydroxide or low-grade magnesium oxide mining by-product (LG-MgO). Raw material supply covered from 56% to 98% of total environmental impacts across all fertilisers, except for ENERGEO CV - BAU and ENERGEO CV TOP - BAU, where core processes were the main contributors to specific sub-categories. LITHOZINC - CE and PHEOSCOR - CE showed similar or enhanced performances compared to the corresponding BAU formulations, indicating potential benefits in employing LG-MgO as precipitant. The supply chains of raw materials exhibited a medium-high social risk in the assessed categories, with Egyptian phosphorite extraction posing the greatest concerns for workers and local communities, and the innovative solution potentially improving the social performance of the fertiliser manufacturing process.

6.1 INTRODUCTION

Phosphorus (P) plays a pivotal role in human society, with around 89% of the global P production being dedicated to agricultural applications, primarily for fertiliser manufacturing (Jupp et al., 2021; Meng et al., 2019). However, the increasing demand for phosphate rocks (PR), driven by intensive agriculture practices, population growth and rapid urbanisation caused the overexploitation and quality decline of available PR reserves (Bacelo et al., 2020; Ryszko et al., 2023). Given this context, recovering P from secondary and renewable sources became crucial to meet the future demand and face the rising social and environmental challenges (Carrillo et al., 2024).

A promising category of secondary and renewable P sources includes P-rich waste streams, such as food waste, crop residues, slaughterhouse waste, livestock manure and municipal wastewater (WW) (Meng et al., 2019; Witek-Krowiak et al., 2022). Among them, WW treatment by-products - such as aqueous phases (i.e., WW treatment effluent, sewage sludge (SS) digestion supernatants and thickening/dewatering liquors), sewage sludge (i.e., digested SS, dewatered/dried SS) and sewage sludge ash (SSA) - are particularly significant due to the large volumes produced worldwide (Canziani et al., 2023). Nowadays, more than thirty P recovery technologies exist globally, reflecting the differences in national contexts and drivers in terms of regulations and/or presence of industrial stakeholders interested in the recovered products (Canziani et al., 2023; Desmidt et al., 2015).

Despite the wide availability of technologies, recovered P still accounts for a small fraction of the fertiliser market. Economic viability remains a major barrier, with the cost of P-based recovered products ranging from 3 €/kg P (wet chemical and thermochemical recovery from SSA) to 28 €/kg P (recovery from SS), exceeding the average price of PR (1.1 €/kg P) and conventional P-based fertilisers (i.e., triple superphosphate - TSP - 2.2 €/kg P (*Businessanalytiq*, 2025a; *Businessanalytiq*, 2025b; Egle et al., 2016). Beyond economic challenges, a growing attention is being paid to assessing the environmental impacts of P recovery scenarios. In this context, Life Cycle Assessment (LCA) is a well-recognised methodology for quantifying the environmental sustainability of products and technologies in the circular economy framework (Del Borghi et al., 2020; Moreschi et al., 2024).

Several studies evaluated the environmental performance of P recovery technologies in comparison with the production of PR-based fertilisers (i.e., TSP) or conventional SS management strategies (i.e., incineration, landfilling, direct land application). Amann et al. (2018) evaluated the overall environmental performances of 18 P recovery technologies, identifying SSA thermochemical treatment as the best trade-off option in terms of heavy metal decontamination, emissions and energy demand, P recovery efficiency and total absence of organic micropollutants in the recovered product. Similarly, Lam et al. (2022) outlined that replacing half of conventional PR-based fertilisers with WW-derived P-based products could reduce the environmental impact of the assessed crop production systems, particularly for the recovery pathways involving struvite precipitation from digester supernatants and SSA thermochemical treatment. On the contrary, Pradel & Aissani (2019) found substantially higher environmental impacts for SS-based fertiliser production compared to TSP, mainly due to lower P concentration, higher chemical, energy and infrastructures requirements, lower fertilizing value of the recovered product and higher gaseous emissions compared to the reference system. Other studies highlighted the potential environmental benefits of alternative sludge valorisation routes, such as energy recovery from hydrochar from hydrothermal carbonisation instead of its direct agricultural application or chemical P recovery (Behjat et al., 2024; Mannarino et al., 2022)

While the environmental impact of existing P recovery technologies was extensively investigated, research on the social and socio-economic implications of P-recovered products across their entire life cycle remains limited. Indeed, existing works mostly address social aspects within broader and integrated economic, environmental, social and technical assessments of different SS management strategies. For example, Ronda et al. (2023) combined technical, socio-economic and envi-ronmental indicators to assess the sustainability of SS thermal valorisation routes (i.e.: flash and slow pyrolysis, gasification and combustion), identifying scalability and heavy metals content in SS treatment by-products as key limiting factors of such strategies. Similarly, Law & Pagilla (2021) applied a Triple Bottom Line (TBL) framework based on economic, social, and environmental factors to compare different P recovery options in the Chicago metropolitan area, concluding that P recovery from SSA as calcium phosphate (LEACHPhos process) and subsequent resource storage in dedicated monofils could offer long-term sustainability benefits, safer sludge disposal option, reduced Operating Expense and an improved recovery efficiency compared with other recovery routes. Only a limited number of studies have explicitly focused on social sustainability. For instance, Teah et al. (2017) evaluated the social performances of fertilisers production and consumption in Japan using mineral (i.e.: imported PR) or secondary sources (WW-derived struvite and hydroxyapatite), demonstrating that PR substitution with recovered P could reduce social impacts of Japanese fertilisers' supply chain, albeit with limitations given by national recycling capacity.

Despite the consistent availability of sustainability assessment of SS management strategies, no studies specifically evaluated the combined environmental and social implications of introducing recovered P into existing, market-ready fertiliser formulations. Specifically, the social impacts associated with the extraction and supply of both primary and secondary raw materials remain largely unexplored within a life cycle perspective. To bridge this gap, this study aims to evaluate the environmental impacts related to the production of four commercial granular fertilisers widely used in Italian professional agriculture for arable and fruit crops, and to assess the social impacts of PR extraction and refinement and key elements (i.e., Mg, S, Ca, Cl) supply for fertiliser production. Environmental Life Cycle Assessment (E-LCA) and Social Hotspot Analysis (SHA) were conducted for the purpose, comparing the performances of a conventional production process, solely based on non-renewable mineral P sources (i.e., phosphorite) with an innovative P recovery technology developed within the ERA-MIN3 EU PHOSTER project, in which a fraction of the mineral source is substituted with P-based products recovered via a wet chemical P recovery process from SSA. A cradle-to-gate perspective was adopted in E-LCA study, covering upstream and core processes. E-LCA focused on specific environmental impact categories, while S-LCA considered proper social impact categories for specific stakeholders.

By integrating environmental and social sustainability assessment, this work provides a comprehensive evaluation of P recovery sustainability, offering novel insights to decision-makers for the transition toward circular and sustainable P management and guiding future research on impact reduction strategies for P recovery technologies.

6.2 MATERIALS AND METHODS

6.2.1 Phosphorus recovery from sewage sludge ash: the PHOSTER project

The PHOSTER project delivered a sustainable circular economy solution for the recovery of secondary minerals and metals from SSA and mining industry by-products to substitute critical raw materials (i.e., P, Mg) in fertiliser manufacturing. Within the project, a wet chemical P recovery process from SSA was developed, comprising SSA wet acid leaching by H_2SO_4 or HCl, leachate coagulation and filtration, and P alkaline precipitation from the P-rich solution by $Ca(OH)_2$ or a low-grade MgO by-product of magnesite calcination (LG-MgO). Process parameters, material and energy inputs, and output flows were directly derived from experimental pilot-scale data reported in Esposito et al. (2024). In that study, the SSA used for process optimisation corresponds to sample S5 previously characterised in Boniardi et al. (2021). Therefore, SSA properties, leaching behaviour and P recovery performance adopted in the present study are based on laboratory-verified and experimentally characterised material,

ensuring full consistency between the recovery process modelling and the underlying ash characteristics.
A simplified process scheme is reported in **Errore. L'origine riferimento non è stata trovata.**.

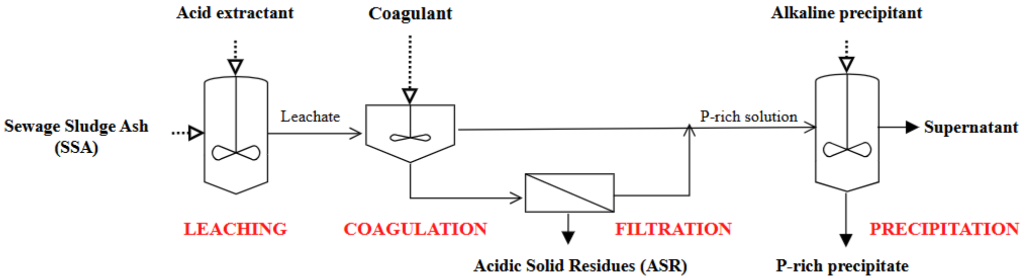


Figure 6-1: *Wet chemical P recovery process from SSA as developed within the PHOSTER project (modified from Esposito et al. (2024)).*

6.2.2 Production of fertilisers and description of analysed scenarios

The four considered fertilisers, named as ENERGEO CV, ENERGEO CV TOP, LITHOZINC and PHEOSCOR, are produced at the Ripalta Arpina plant (Cremona, Italy) of TIMAC AGRO Italia S.p.A., which produces and packages different products, as simple phosphate mineral fertilisers, NPK compound minerals and NPK organo-minerals. The four fertilizers were selected based on their extensive use in Italian professional agriculture for arable and fruit crops, representing approximately 10% of the total granular fertiliser production in Italy in 2024 by the company (**Table C-1** in Appendix C**)**. The manufacturing of such products takes distinctly place in a superphosphate production plant (semifinished product), a nitrogen melting plant (semifinished product) and a granulation plant. The superphosphate production plant comprises a H_2SO_4 storage and mixing unit, a raw phosphorite grinding unit and a diluted H_2SO_4 and ground phosphorite mixing unit. The latter is paired to a wet washing unit, to abate gases generated by the exothermic reaction, and two drying plants, equipped with bag filtration. More details about the discussed plant operation are provided in El Chami et al. (2023). Two main scenarios were outlined: a Business-As-Usual (BAU) scenario, in which fertilisers are solely based on imported PR (i.e., phosphorite), and a Circular Economy (CE) scenario, in which a fraction of PR is replaced with P-based products recovered via the wet chemical P recovery process from SSA developed within the PHOSTER project. A detailed description of the CE scenario is reported in **Tables C2- C4** (Appendix C).

6.2.3 Environmental Life Cycle Assessment and Social Hotspot Analysis

6.2.3.1 Goal and scope definition

E-LCA was conducted to evaluate the environmental impacts related to the production of four granular fertilisers (ENERGEO CV, ENERGEO CV TOP, LITHOZINC, PHEOSCOR). The analysed product systems belong to the Central Product Classification (CPC) group 346 (Fertilisers and pesticides) of the UNSD - CPC classification. The product system function is the production of granular NPK fertilisers via a granulation process.
The analysed product systems belong to the CPC class 1611 (Natural calcium phosphates, natural aluminium calcium phosphates and phosphatic chalk) of the UNSD - CPC classification. The product system function comprises P production via a conventional mining or innovative recovery process from SSA.
The software employed in this study was SimaPro version 10.2 relying upon the Ecoinvent database (v 3.11). The E-LCA was conducted based on the ISO 14040:2006 and ISO 14044:2006 standards. The product-specific requirements indicated in Product Category Rule (PCR) 2010:20 Version 2.21 (Mineral or chemical fertilisers), reference version of EPD S-P-01960, were followed. Based on the product life-cycle subdivision proposed in the reference CPC, only upstream processes (from cradle-to-gate) and core processes (from gate-to-gate) were considered, therefore a “cradle-to-gate” analysis was performed. The functional unit was 1 ton of produced fertiliser, including the packaging of the final product.

E-LCA adopts a cradle-to-gate perspective, encompassing upstream and core processes. The use phase was not included in the assessment.
Upstream processes cover:

- the production and transport of raw materials for granulation,
- the production of primary and secondary packaging of fertilisers,
- the production of fossil fuels for the manufacturing plant.

Conversely, core processes encompass:

- the transport of primary and secondary packaging transport,
- fertiliser manufacturing,
- water consumption,
- thermal and electric energy generation,
- waste management.

However, the production of raw material packaging and bag filters was not considered.
SHA was conducted to evaluate the social impacts of P extraction and refinement from phosphorite rocks and those of key raw materials supply for

fertiliser production (i.e., Mg, S, Ca, Cl), then preliminary investigation of representative companies illustrates how the generic risk profiles manifest in actual operations. Conversely to E-LCA, the social assessment was framed as an exploratory hotspot analysis focused on the upstream phase (raw material supply), with the primary aim of identifying and illustrating the most critical social risks along the raw materials supply chains (Mancini et al., 2023). This methodological choice is in agreement with relevant scientific literature, since it was pointed out that the raw material extraction and upstream processes typically represent major social hotspots in agri-food and fertilizer supply chains while the downstream production stages can substantially influence final social outcomes, depending on location-specific labor conditions and manufacturing practices (Tsalidis et al., 2020).

6.2.3.2 Life Cycle Inventory

The Environmental-Life Cycle Inventory (E-LCI) is the LCA step involving the collection of data and the application of calculation procedures to quantify inputs and outputs from a product system (*EC*, 2010).

Data for fertiliser manufacturing were derived from the environmental product declaration (EPD) released to TIMAC AGRO Italia S.p.A. in 2018 (*EPD - TIMAC AGRO Italia S.p.A.*, 2018). The EPD was conducted according to the reference PCR, as mentioned in the previous subsection, and data contributing less than 1% of the total mass and energy flows were excluded from the E-LCI.

Whenever possible, the production system was divided into sub-processes and data were directly collected for each one. When this approach was not feasible, impacts were allocated according to the relative mass flow rates of each product. This applies to shared flows recorded at aggregated plant level rather than per product (e.g., site-level energy), for which product-specific sub-metering was not available. In those cases, flows were allocated to co-produced fertilisers based on relative mass outputs, consistently with the underlying EPD/PCR approach. The collected data were divided into input ones, referred to materials, transport and energy, and output ones, referred to generated products and polluting streams released in the receiving compartments (i.e., air, water and soil). Data were also classified as specific data, collected from direct measurements or obtained from target literature, generic data, obtained from specific databases, and proxy data, based on estimations or average values. Whenever possible, specific data were employed. Upstream processes primarily relied on specific data. For BAU scenario, information on fertiliser manufacturing, energy consumption and gaseous emissions was sourced collected from 2024 company reports. Specifically, for products already covered by EPD (i.e., LITHOZINC NPK 6-12-16 Bulk), data from the preparatory E-LCA to the EPD were used. Data for the formulations related to CE scenario were provided by the Department of Civil and Environmental Engineering (DICA) of Politecnico di Milano, as deliverables of the PHOSTER project, and refer to recovery configurations and products whose chemical composition and regulatory compliance were previously experimentally validated. Data related to raw materials transportation were calculated based on the means of transport and the supposed distance between raw-materials supplier sites and fertiliser manufacturing plant. Other data related to raw materials, fuels and electricity were collected from the Ecoinvent database (v. 311). Proxy data were employed for a limited number of raw materials and did not significantly contribute (≤ 10%) to the total impact in each considered sub-category. All data were geographically referred to Europe.

As previously mentioned, upstream processes primarily include the manufacturing and transport of raw materials for granulation, generation of primary and secondary packaging of fertilisers and of fossil fuels for the production plant. The type and quantity of required raw materials to produce the four fertilisers under both scenarios are reported in **Tables C-4** and **C-5** (Appendix C). It should be noticed that the total fraction of raw materials in each formulation exceeds 100%, as part of the mineral wastes are recirculated within the production process.

Primary and secondary packaging data are provided in **Table C-6** (Appendix c), assuming identical quantities required in BAU and CE scenarios. Plant electric and thermal energy demands are satisfied through four boilers (B) and one cogeneration (C) units fed with natural gas, whose annual consumption is reported in **Table C-7** (Appendix C). In addition, an annual diesel consumption of 124,149 L for transport was estimated.

Core processes cover the transport of primary and secondary packaging, fertiliser manufacturing, plant water consumption, thermal and electric energy generation and waste management. Truck transport was assumed for the shipment of primary and secondary packaging. Natural gas consumption was converted to primary energy values using a lower heating value of 9.7925 kWh/Std m^3. A thermal efficiency of 42.4% was assumed for the cogeneration unit. This unit provides only part of the plant electricity demand, while the rest is purchased from the grid. Data of electricity purchase, sale and production are provided in **Table C-8** (Appendix C).). An electric efficiency of 36.9% was calculated based on the net electricity production. 22.5% of the plant total energy consumption is related to superphosphate and super-calcium-nitrogen powdering, while the remaining is associated to granulation.

Emissions from the cogeneration unit were allocated between electric (65.2%) and thermal energy production (34.8%), following the PCR 2007:08 (Electricity, steam and hot/cold water generation and distribution) and applying the allocation factors listed in **Table C-9** (Appendix C).). Process-specific electricity uses of 12 kWh/tonne for powdered products (superphosphate, super-calcium-nitrogen) and 23 kWh/tonne for granular fertilisers were provided by TIMAC AGRO Italia S.p.A. Moreover, the company registered a groundwater consumption of 8529 m^3 and 22526 m^3 for superphosphate and NPK-fertiliser production, respectively. Process water was supposed to totally disperse as vapor, since the moisture of final products and raw materials is comparable. No water discharges were recorded in the reference EPD. Plant diesel consumption was estimated assuming a diesel density of 0.84 kg/L and a lower heating value of 42.877 MJ/kg, resulting into a total thermal energy consumption of 4,471,435 MJ. Plant gaseous emissions and waste streams are reported in **Tables C-10** and **C-11** (Appendix C).), respectively. No wastes are generated by the production process, as mineral wastes are recirculated back to the production process.

The SHA followed ISO 14075:2024 by employing the Social Hotspots Database (SHDB) as the primary data source, providing standardized characterization factors derived from multi-regional input-output analysis across 140+ countries and 57 economic sectors (Benoit-Norris et al., 2012). This approach ensures consistent assessment of direct, indirect, and induced social impacts across the entire supply chain. Quantity and criticality of raw materials, as defined by the European Commission, were used as cut-off criteria to detect main potential impacts and social hotspots. Whenever possible, the system was divided into sub-processes and data were directly acquired for each sub-process. Whenever not possible, no allocation rule was applied, and the same social performance factors were attributed to each product. Specific data were gathered from scientific publications and reports of main raw material suppliers and main producing countries (e.g.: USGS Mineral Commodity Summaries 2024 - USGS, 2024).

Firstly, to identify the main social hotspots, the analysis was extended to the main producing countries of the raw materials under investigation, such as P (China, USA, Morocco, Egypt), Mg (Brazil, China, Israel, Kazakhstan), S (China, Russia, Saudi Arabia, USA), Ca (Germany, Malaysia, UK, Zambia) and Cl (Canada, France, Thailand, USA). By comparing the social risk index of different countries, the geographic areas with major social criticises associated to raw material supply were identified. In case the social conditions of assessed countries were unknown, the world’s largest producers for the specific material were considered. Consequently, the average social risk values associated with the supply chains of each raw material were weighted based on the production data of each considered country. A risk rating scale was employed to score the outcomes from 1 (low risk) to 4 (very high risk) highlighting the most critical social hotspots.

6.2.3.3 Life Cycle Impact Assessment

This phase translates the elementary flows gathered in the inventory into impact indicators related to human health, natural environment and resource depletion. For the E-LCA, in accordance with the ISO standards 14040-14044 and in agreement with the scientific literature of the sector, environmental impacts were assessed using the Environmental Footprint 3.1 impact assessment methodology, with a subset of relevant impact categories was selected according to their importance on the contribution to the overall normalized single-scores in EF3.1. (Amann et al., 2018; Bassi et al., 2023; Behjat et al., 2024; El Chami et al., 2023; Lam et al., 2022; Mannarino et al., 2022; Pradel & Aissani, 2019; Smol et al., 2020). The impact categories and sub-categories, related acronyms and units of measures are listed in **Table 6-1**, with the chosen subset highlighted in bold.

Lastly, to strengthen the discussion of the results and provide a broader perspective, comparisons were made with relevant scientific literature in the field (El Chami et al., 2023).

Table 6-1: Impact categories and related sub-categories accounted in the E-LCA.

Impact category and sub-category	Acronym	U.M.
Environmental impact potentials		
Global Warming Potential fossil	GWPfos	[kg CO_2 eq]
Global Warming Potential biogenic	GWPbio	[kg CO_2 eq]
Global Warming Potential land use and land use change	GWPlul	[kg CO_2 eq]
Global Warming Potential total	GWPtot	[kg CO_2 eq]
Depletion Potential of the stratospheric ozone layer	ODP	[kg CFC-11 eq]
Acidification Potential	AP	[mol H^+ eq]

Eutrophication Potential - freshwater	EPfw	[kg P eq]
Eutrophication Potential - marine	EPma	[kg N eq]
Eutrophication Potential - terrestrial	EPte	[mol N eq]
Photochemical ozone creation potential	POCP	[kg NMVOC eq]
Abiotic depletion potential for minerals and metals	ADPm	[kg Sb eq]
Abiotic depletion potential for fossil resources	ADPf	[MJ - net calorific value]
Water deprivation potential	WDP	[m^{3}depriv]
Resource use		
Renewable primary resources used as energy carrier (fuel)	PERE	[MJ]
Renewable primary resources with energy content used as material	PERM	[MJ]
Total use of renewable primary energy	PERT	[MJ]
Non-renewable primary resources used as an energy carrier (fuel)	PENRE	[MJ]
Non-renewable primary resources with energy content used as material	PENRM	[MJ]
Total use of non-renewable primary energy resource	PENRT	[MJ]
Secondary materials	SM	[kg]
Renewable secondary fuels	RSF	[MJ]
Non-renewable secondary fuels	NRSF	[MJ]
Net use of fresh water	FW	[m^3]
Land use related impacts	LU	[-]
Wastes disposal		
Hazardous waste disposed	HWD	[kg]
Non-hazardous waste disposed	NHWD	[kg]
Radioactive waste disposed	RWD	[kg]
Human health		
Particulate matter emissions	PM	[disease inc.]
Ionizing radiation, human health	IR	[kBq U235 eq]
Eco-toxicity (freshwater)	ET	[CTUh]
Human toxicity (cancer effects)	HTc	[CTUh]
Human toxicity (non -cancer effects)	HTnc	[CTUh]

The three waste-related indicators reported in **Table 6-1** (HWD, NHWD and RWD, in kg) are not part of EF3.1; they are additional waste indicators sourced from the EDIP 2003 set.

The SHA focused on two key stakeholder categories: workers and local communities. These categories were considered as the most directly affected groups along upstream extraction and refining stages of the key raw materials under investigation (e.g., P, Mg, S, Ca, Cl). Stakeholder sub-categories and inventory indicators were selected as a screening set, prioritising relevance for mining/smelting supply chains. Workers were defined as the personnel involved in the extraction and refinement of raw materials for fertiliser manufacturing. This category was furtherly subdivided into the following sub-categories: health and safety, wages, social benefits, working conditions, discrimination and freedom of association and collective bargaining. Local communities were defined as communities and organisations relying on natural and artificial local resources, having a mutual interest in protecting and improving their quantity and quality. This category included the sub-categories of local employment and cultural heritage & land rights sub-categories. Social impact categories and sub-categories are listed in **Table 6-2**.

Table 6-2: Selected social impact categories (bold) and sub-categories accounted in the SHA.

Right to decent work	Wage assessment Forced labor Freedom of association
Health and safety	Workplace accidents
Society	Gender equality
	Poorness and inequality
Governance	Corruption
Community	Access to safe drinking water
	Access to sanitary facilities
	Property rights

Representative supplier companies were then selected to illustrate how the sector level social risk profiles identified in the SHA can materialise at facility level in concrete operational settings. In this way, company specific information on policies, certifications and reported practices complements the aggregated database results, providing qualitative insight into how generic country-sector risks translate into actual working conditions and community interactions for key suppliers in the fertiliser value chain. Consequently, a semi-quantitative approach was adopted using a reference-scale scoring based on available evidence and SHDB risk information. This approach converts social inventory outcomes into a score based on established reference scales defined according to national/international standards or best practices. This approach is in line with the Guidelines for Social Life Cycle Assessment of Products and Organisations (UNEP, 2020), and it provides social performance results. Specifically, a suggested reference scale ranges from - 2 to + 2, where score +2 indicates an “Ideal Performance”: a positive output is achieved and accurately reported, for instance the compliance with ideal conditions for a specific subcategory, according to the International Labour Organisation (ILO) conventions (such as Freedom of Association and Protection of the Right to Organise Convention (1948/87), Right to Organise and Collective Bargaining

Convention (1949/98), Forced Labour Convention (1930/29), Equal Remuneration Convention (1951/100), Discrimination (Employment and Occupation) Convention (1958/111), Occupational Safety and Health Convention (1981/155)). Conversely, value -2 indicates a "non-compliance" in a sub-category where a high-risk social issue is detected and no action taken (-2 where explicitly mentioned may also indicate lack of data). The score +1 indicates a "progress beyond compliance" so company proactive behaviour positively exceeding the regulatory compliance, while - 1 indicates "slightly below compliance level" with one or more non-conformities in a social category although it does not represent a social hotspot for a specific sub-category. Lastly, the score 0 represents "Compliance" with local international laws.

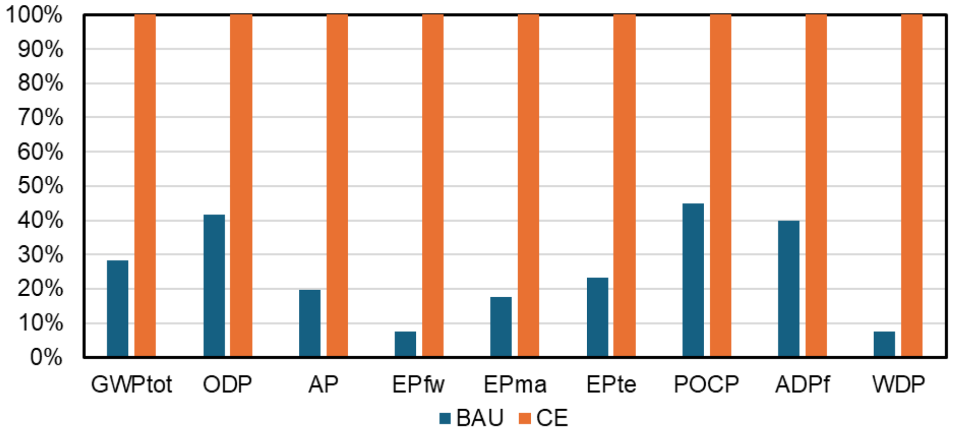
This methodological approach allows linking sector level social risk profiles to actual industrial actors in the investigated value chain, using publicly available information on certifications, policies and reported practices to illustrate how generic country-sector risks can manifest - or be mitigated - at specific sites. It should be noted that the companies are not intended to represent the full variability of their sectors, but rather to exemplify "better practice" suppliers within the BAU value chain vs. CE scenario. Consequently, the aggregated SHA based on the social database and input-output information remains the primary tool for identifying and comparing social hotspots across materials and countries, while the analysis of representative companies is explicitly framed as a qualitative, illustrative step that helps interpret how the macrolevel risk patterns may translate into concrete working and community conditions for key suppliers.

6.3 RESULTS AND DISCUSSION

6.3.1 Environmental Life Cycle Impact Assessment

6.3.1.1 Total impacts

Figure 6-2 reports the set of impact categories (GWPtot, ODP, AP, EPfw, EPma, EPte, POCP, ADPf and WDP) as highlighted in section 6.2.3.3. For clarity and brevity, each fertiliser assessed in a specific scenario is referred to as "fertiliser - scenario" and, throughout the manuscript, the fertilisers are identified using the following abbreviations: F1 (ENERGEO CV), F2 (ENERGEO CV TOP), F3 (LITHOZINC) and F4 (PHEOSCOR). Detailed numerical results for all assessed impact categories are reported in **Tables C-12-C-15** (Appendix C).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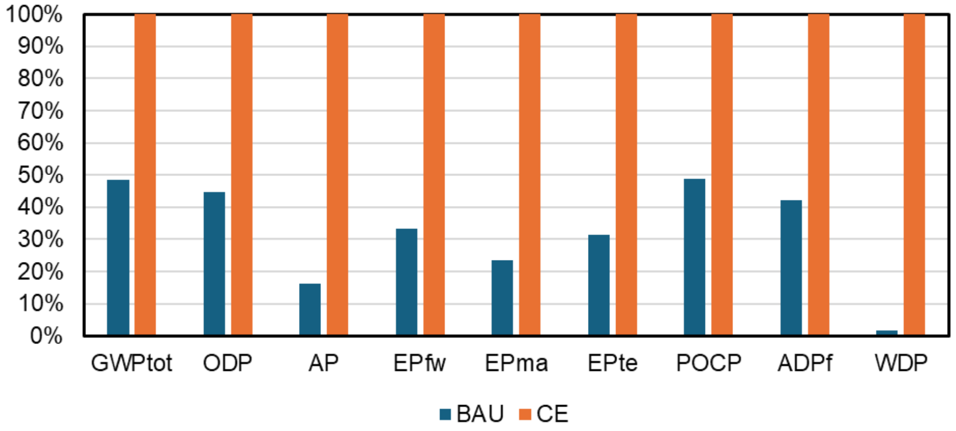
■ BAU ■ CE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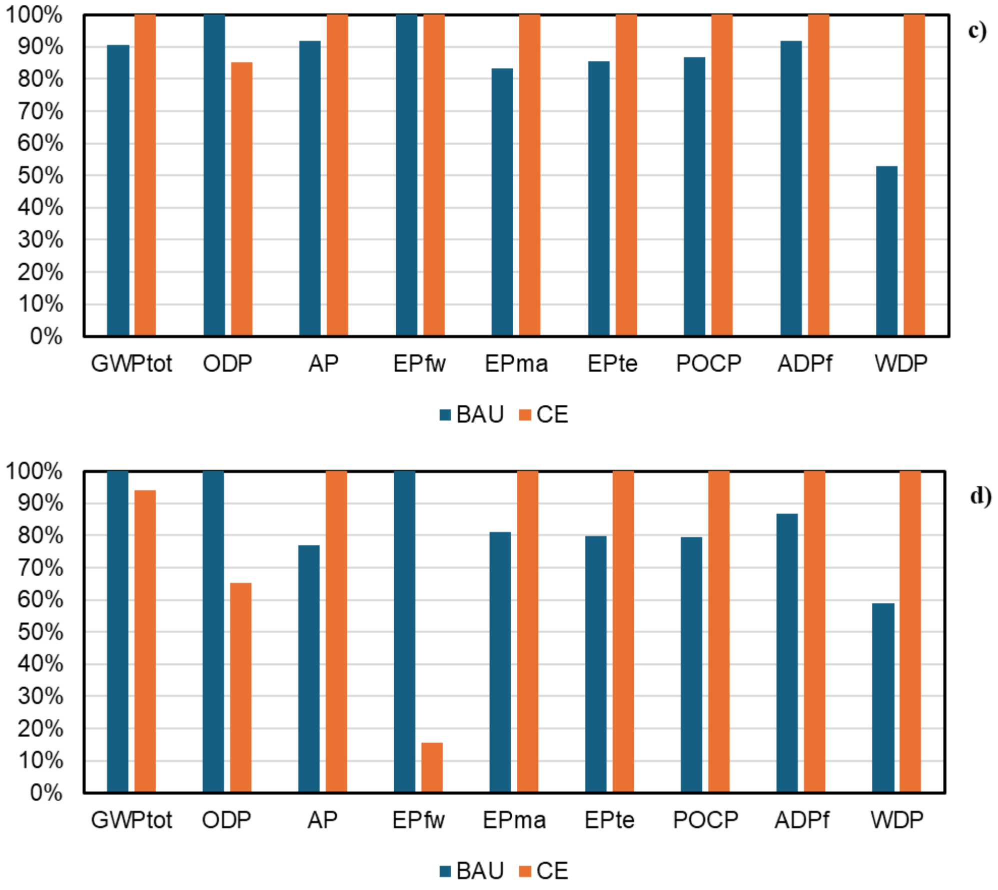

Figure 6-2: Total environmental impacts associated with F1 (a), F2 (b), F3 (c) and F4 (d) production assessed in BAU and CE scenarios. Values are individually normalised to the higher impact between BAU and CE scenarios. Note: GWPtot: Global Warming Potential total; ODP: Ozone Depletion Potential; AP: Acidification Potential; EPfw: Eutrophication Potential freshwater; EPma: Eutrophication Potential marine; EPte: Eutrophication Potential terrestrial: POCP: Photochemical Ozone Creation Potential; ADPf: Abiotic Depletion fossil and WDP: Water Depletion Potential.

Overall, the environmental benefit of CE formulations remained limited when considering the subset of impact categories reported in **Figure 6-2**, with CE generally underperforming BAU for AP, EPma, EPte, POCP and WDP across the four fertilisers. This behaviour is consistent with previous work showing increased burdens for several air and water related categories when comparing

P recovery routes from SSA with conventional TSP production through the PolFerAsh process (Smol et al. 2020). On the contrary, other studies (Amann et al., 2018; Lam et al., 2022) indicated improvements in specific impact sub-categories when P recovery scenarios were compared to scenarios comprising SS conventional management (i.e., incineration, landfilling and direct land application) or mineral-based fertiliser production. Amann et al. (2018) revealed that SSA-based recovery technologies achieved comparable or reduced impacts in cumulative energy demand, GWPtot and AP relative to the adopted reference system. These benefits were attributed to credits for mineral-P substitution and high P recovery efficiency (70% - 90% of influent P in WW treatment plant), which offset the increase in energy and chemicals inputs due to P recovery process. Similarly, Lam et al. (2022) reported that SSA thermochemical treatment (RP3) and wet acid extraction (RP6) generally reduced EPfw and AP, while potential benefits in ET depend on the trade-off between heavy metals content in the final product and credits from biogas production.

$Ca(OH)_2$based fertilisers (F1 and F2; **Figure 6-2a-b**) showed higher impacts in CE than in BAU for most of the displayed categories, whereas LGMgO-based fertilisers (F3, F4; **Figure 6-2c-d**) exhibited a more favourable pattern, with CE leading to reduced impacts in several climate and resource related categories (e.g. GWPtot, ODP, EPfw) while still performing worse for AP, EPma, EPte and POCP. These results confirm the potential environmental advantage of replacing conventional $Ca(OH)_2$ with waste-derived LGMgO in wet chemical P recovery processes, although trade-offs among impact categories remain.

The comparison between HCl- and H_2SO_4-based configurations indicates that H_2SO_4-based formulations (F2, F4) tend to reduce some of the reported impact categories relative to their HCl-based counterparts (F1, F3), suggesting an influence of the acid extractant on the overall environmental performance of the fertilisers.

The higher burdens observed for CE formulations in several impact categories are primarily driven by the additional chemicals and energy required by the SSA-based P recovery process (e.g. production of H_2SO_4/HCl and $Ca(OH)_2$ or LG-MgO, thermal treatments, pumping and mixing), which offset the credits from mineral P substitution under the current efficiency levels. In particular, the contribution analysis shows that acid production, alkali supply and electricity for the recovery line are key hotspots in the CE configurations, consistent with previous LCA studies on P recovery from SSA and sludge-derived streams.

Conversely, the reduced impacts of LG-MgO-based formulations in climate- and resource-related categories suggest that using waste-derived precipitants with lower embodied energy and fossil resource demand can partially compensate the additional burdens from the recovery process. These patterns highlight that improving the environmental performance of CE options is less a matter of "using SSA-derived P per se" (Lam et al., 2022)than of optimising the specific recovery route (chemicals, energy and integration with existing plants), which aligns with recent findings on the sensitivity of P recovery systems to reagent origin and process energy efficiency (Witek-Krowiak et al., 2022).

Finally, the environmental impact potentials for the four assessed P recovery configurations from SSA were compared (**Table C-16** in Appendix C, **Figure 6-3**). LG-MgO-based configurations generally outperformed $Ca(OH)_2$-based ones, except for EPma, EPte and WDP. This evidence might be attributed to the absence of $Ca(OH)_2$, whose production represented one of the highest contributors to GWPtot. No significative differences were noticed between the environmental performances of HCl- and H_2SO_4-based configurations.

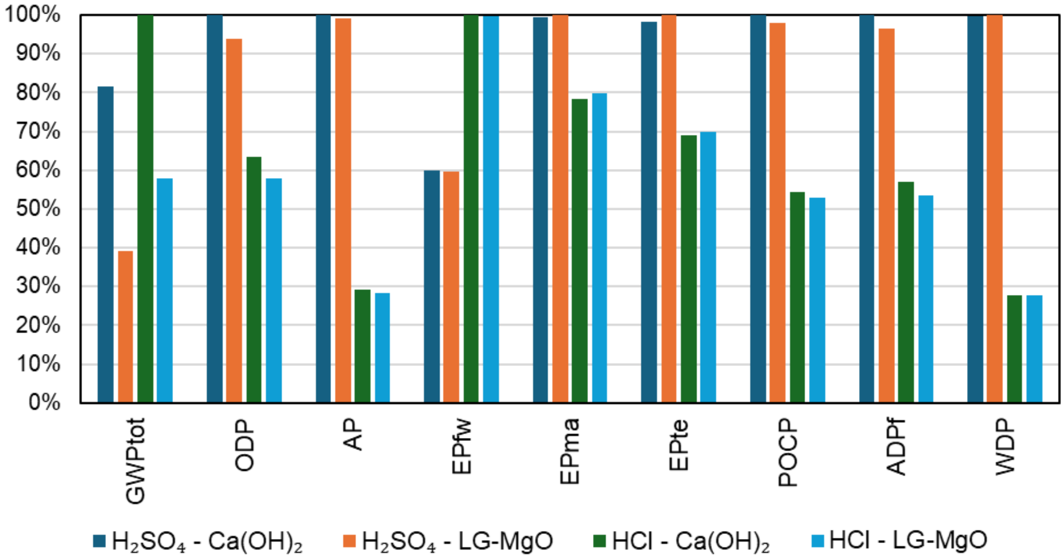

Figure 6-3: Total environmental impacts of the four assessed configurations of P recovery from SSA. Impacts are individually normalised to the higher value of each sub-category. Note: GWPtot: Global Warming Potential total; ODP: Ozone Depletion Potential; AP: Acidification Potential; EPfw: Eutrophication Potential freshwater; EPma: Eutrophication Potential marine; EPte: Eutrophication Potential terrestrial: POCP: Photochemical Ozone Creation Potential; ADPf: Abiotic Depletion fossil and WDP: Water Depletion Potential.

6.3.1.2 Upstream and core impacts

Figure C-1 (SM) displays total impacts distribution across upstream and core phases for each fertiliser and all investigated scenarios. Upstream phase always exhibited a major impact contribution on the production of each analysed fertiliser, except for F1 - BAU and F2 - BAU. Specifically, the environmental impacts from raw materials production ranged between +55% and +98% for almost all sub-categories. Exceptions were noticed for specific sub-categories in case of F1 - BAU (GWPfos, GWPtot, EPma, EPte, POCP and HWD), F2 - BAU (EPma, POCP and HWD), F3 - CE (HWD) and F4 - CE (HWD), with core contributions ranging between +60% and +72% of total impacts.

Focusing on GWPtot, the major upstream contributions across each scenario derived from phosphorite mining, P-based products manufacturing (e.g.: SSP complex, DAP, TOP PHOS) and P recovery from SSA (i.e., production of HCl (7% - 22%), H_2SO_4 (8% - 11%) and $Ca(OH)_2$ (28% - 32%). Conversely, the main core contributions were represented by thermal (35% - 47%) and electric (13% - 17%) energy generation. In line with these findings, previous studies identified raw material manufacturing as the major environmental hotspots for fertilisers containing recovered P, with significant impacts deriving from the production of chemicals used in P recovery from SSA (H_2SO_4, HCl, H_3PO_4, HNO_3, CaO, $CaCO_3$, NaOH) or from aqueous phases (NaOH, $MgCl_2$, NH_4Cl) (Amann et al., 2018; Behjat et al., 2024; El Chami et al., 2023; Mannarino et al., 2022). Other relevant contributors to the environmental footprint as reported in the scientific literature include water consumption and thermal and electric energy use (Behjat et al., 2024; Mannarino et al., 2022; Pradel & Aissani, 2019).

When examining changes in impact distribution between upstream and core phases, a shift from $Ca(OH)_2$-based to LG-MgO-based fertilisers in BAU scenario generally increased upstream contribution to total impacts (+10%), whereas marginal variations were ascribed to this change in CE scenario. Furthermore, no relevant displacements were noticed in upstream or core phases relevance when changing the acid extractant. Overall, that the transition from BAU to CE scenario increased (+10% - +15%) upstream phase relevance for F1 and F2, while no significant changes were observed for F3 and F4 in the analysed sub-categories.

6.3.2 Comparison between E-LCA and scientific literature outcomes

To enhance discussion on findings from current work, a comparison with the E-LCA study recently performed by El Chami et al. (2023) was conducted, assessing the environmental impacts of producing four different fertilisers (named as TIMATECH, EUROCOD, PRIME, MAGNIFIQUE) under two scenarios: a baseline scenario (BASE), where fertilisers are solely based on commercial-grade inputs, and a circular scenario (CIR), where conventional N source (i.e., Egyptian urea) was partially replaced with $(NH_4)_2SO_4$ recovered from by-products of plastic industry (i.e., SO_2).

The study from El Chami et al. (2023) was selected for comparison due to its close alignment with the objectives of the present work, namely the environmental impact assessment of the partial substitution of mineral-based resources with secondary products in market-ready fertiliser formulations. Despite this, it is important to acknowledge that the two studies differ in key methodological aspects and system boundary definitions. Specifically, El Chami et al. (2023) adopted the Environmental Footprint 3.0 method and included the use phase of the fertilisers within the system boundaries (downstream phase), while the present study relies on the Environmental Footprint 3.1 method and is limited to upstream and core phases. Such differences are known to considerably influence E-LCA outcomes and, in turn, limit the possibility to drawn unbiased quantitative comparisons. For this reason, the comparison presented in this section is intended to be qualitative rather than conclusive, aiming to relate the present findings within the existing literature while explicitly accounting for methodological heterogeneity.

For consistency with the labelling approach adopted in sub-section 6.3.1.1, each fertiliser assessed in a specific scenario is referred to as "fertiliser - scenario" and, throughout the manuscript, the fertilisers are identified using the following abbreviations: C1 (TIMATECH), C2 (EUROCOD), C3 (PRIME) and C4 (MAGNIFIQUE).

Figure C-2 (SM) show the variations in specific impact sub-categories (GWPtot, AP, EPfw, ET, LU, WDP, ADPf) in response to a change from conventional (BAU or BASE) to innovative (CE or CIR) formulations. The partial substitution of N source resulted in different outcomes, depending on the considered impact sub-category and fertiliser. C1 was the only fertiliser proving environmental benefits across each impact sub-category investigated, with the most significant reduction in WDP (-16.1%). This could be attributed to the lower N content in both C1 - BASE and C1 - CIR compared to other formulations (**Table C-17** in Appendix C). GWPtot (-4.4% - -9.2%) and EPfw (-2.9% - -5.5%) were the sub-categories consistently reduced across all fertilisers. These reductions were attributed to the shorter transport distance between the new N recovery location (i.e., Milan, Italy) and the fertiliser manufacturing plant (i.e., Ripalta Arpina, Cremona, Italy), which potentially reduce greenhouse gases emission, and to the reduced N content in CIR formulations, which may decrease N emissions from fertiliser application. The innovative formulations investigated in this work showed less significant environmental benefits compared to that of El Chami et al. (2023), with CE underperforming BAU across every fertiliser for AP, Epma, Epte, WDP and ADPf. The environmental impact sub-categories proving impact reduction were GWPtot (F4, -5.8%), EPfw (F3 and F4, -17.7% and -84.3%, respectively) and ET (F4, -27.4%). Conversely, significant increases were noticed for WDP, especially for F1 and F2 (+1226% and +6340%, respectively), indicating the need for further P recovery process optimisation, as already indicated by Boniardi et al. (2024).

Examining upstream and core impacts related to GWPtot (**Figure C-3**, SM), CIR formulations showed a significant decrease in upstream contribution, potentially due to the abovementioned reduction in transport distance. Conversely, no significant variations were noticed in core contribution, as energy management was already optimised through cogeneration (El Chami et al., 2023). In the present work, changing from BAU to CE formulations significantly increased upstream contribution for F1 (+299.6 kg CO_2 eq) and F2 (+182.7 kg CO_2 eq), while a less relevant increase (+53 kg CO_2 eq) and a slight decrease (-12 kg CO_2 eq) were noticed in case of F3 and F4, respectively. As discussed in sub-section 6.3.1.1, the upstream improvement in F4 - CE could be related to the absence of $Ca(OH)_2$, whose production significantly contributed to GWPtot, and to the substitution of HCl with H_2SO_4, whose supply generates fewer gaseous emissions.

6.3.3 Social Hotspot Analysis (SHA)

Figure 6-4 presents the outcomes from the SHA conducted and expressed in terms of average social risk score. As previously mentioned, the average social risk values associated with the supply chains of each raw material were weighted based on the production data of each considered country.

Most of the social risk values ranged between 2 and 4, indicating a medium-high risk for raw material supply chains across the selected social impact sub-categories. Moreover, elevated social risks were identified for phosphorite production in the considered stakeholder categories.

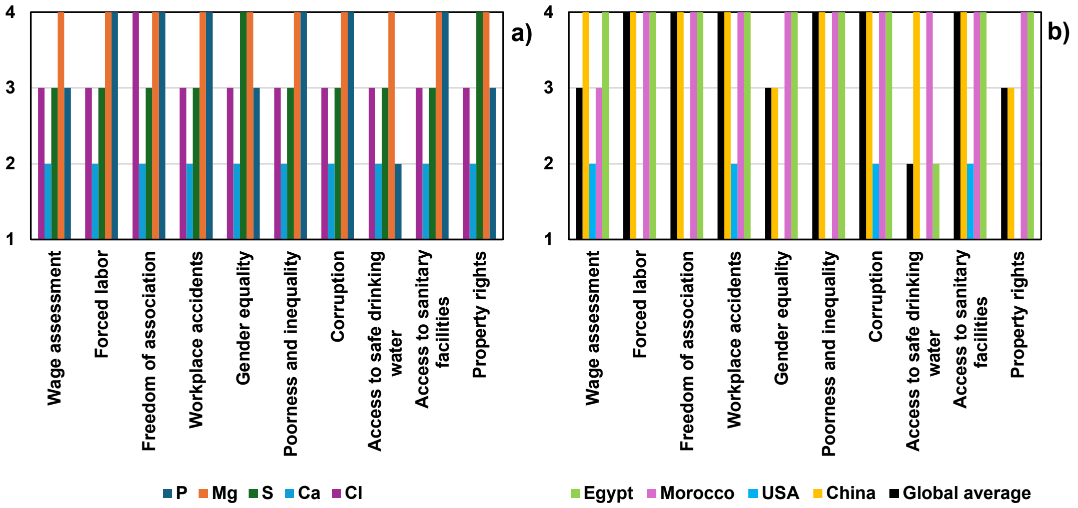

Figure 6-4: Average social risk scores for the supply chains of P, Mg, S, Ca and Cl (a), focus on P extraction and refinement in China, USA, Morocco and Egypt (b). (1 = low, 2 = medium, 3 = high, 4 = very high).

Subsequently, a more detailed analysis of the social performance of main raw materials and chemicals used in BAU (Egyptian phosphorite) and CE (H_2SO_4, HCl, $Ca(OH)_2$, LG-MgO) scenarios, as well as of a hypothetical P recovery plant, was carried on based on available data from the raw materials suppliers' official websites. The relevant data are summarised in **Table C-18** (Appendix C), while the derived social impact scores are displayed in **Table 6-3**.

An Egyptian phosphorite supplier with an annual production of around 5 million tonnes of concentrated PR was assumed as representative supplier. According to its official website, the company declares an explicit commitment to improve freedom of association, reduce workplace accidents and reject forced labour. Therefore, these sub-categories were assigned the social performance value of +1. Despite the supplier specifically affirmed to reject any discrimination form, a value of 0 was assigned to the "gender equality" sub-category, as no explicit mention was found in the company website. A score of +2 was not assigned to any sub-category, due to the absence of third-party verification of company commitments. An Italian H_2SO_4 supplier with an annual output of around 560 ktonnes/year of H_2SO_4 was chosen as sector representative. Since the supplier is certified with SA8000® and ISO 45001:2018 certifications, a score of +2 was assigned to the related sub-categories. Despite the environmental and health risks typically associated with sulfuric acid production, a score of 0 was assigned to remaining sub-categories, as the collected data refers only to official information. The HCl supplier was selected as representative for the HCl production sector in Italy. Since the supplier is certified with ISO 45001:2018 certification, a value of +2 was assigned to the related sub-categories. Moreover, a score of +1 was assigned to sub-categories reflecting company commitment to avoid any discrimination based on age, gender, sexual orientation, health status, ethnicity, nationality, political opinions and religious beliefs.

Table 6-3: Social impact scores for the main P-based raw materials employed in BAU and CE scenarios across every assessed social impact sub-category. The scores are visually represented with a colour scale ranging between red (-2) and green (+2).

		Phosphorite supplier	H_2SO_4 supplier	HCl supplier	$Ca(OH)_2$ supplier	LG-MgO supplier	P recovery plant
Right to decent work	Wage assessment	0	0	0	0	+1	0
	Forced labour	+1	0	0	0	0	0
	Freedom of association	+1	0	0	0	0	0
Health and safety	Workplace accidents	+1	+2	+2	+2	+2	+1
Society	Gender equality	0	+1	+1	0	0	+1
	Poorness and inequality	0	+1	+1	0	+1	0
Governance	Corruption	0	0	0	0	0	0
Community	Access to safe drinking water	0	0	0	0	0	0
	Access to sanitary facilities	0	0	0	0	0	0
	Property rights	0	0	0	0	0	0

The LG-MgO is supplied by the private mining company Magnesitas Navarras (Navarre, Spain), partner of the PHOSTER project. Since the company holds a third-party certification in occupational health and safety, a score of +2 was assigned to the “work accidents” sub-category. A leading Italian producer of calcic lime, dolomitic lime and derived products, with a capacity exceeding 2 million tonnes/year of calcic lime was selected as representative for $Ca(OH)_2$ supplier. Since the supplier holds an ISO 45001:2018 certification, a value of +2 was assigned to the related sub-categories. The P recovery plant was assumed to be located in Italy. The overall performance of the Italian chemical sector was considered for the analysis, assigning a value of +1 to the sub-categories in which this sector performs better than the national average.

Given its upstream focus, the SHA does not capture potential social risks or benefits associated with downstream fertiliser production and use; a cradle-to-gate SLCA could in principle lead to different overall social patterns.

6.4 CONCLUSIONS

This study reveals that replacing mineral phosphorus with SSA-derived phosphorus in commercial granular fertilisers delivers only modest environmental gains under the analysed conditions. Specifically, Circular Economy (CE) formulations frequently underperform Business-As-Usual (BAU) scenarios across key air- and water-related impact categories, including acidification potential (AP), marine eutrophication (EPma), terrestrial eutrophication (EPte), photochemical ozone creation (POCP), and water deprivation (WDP). Notably, CE options employing low-grade magnesium oxide (LG-MgO) as precipitant demonstrate reduced climate- and resource-related burdens relative to calcium hydroxide-based configurations, emphasising the critical influence of precipitant selection in the recovery process. The upstream Social Hotspot Analysis (SHA) identified medium-to-high social risks - particularly associated with Egyptian phosphorite extraction - while indicating that partial substitution with recovered phosphorus could diversify supply chains and reduce risks. These results highlight the environmental and social trade-offs of the process, requiring strategic view to realise sustainable, large-scale SSA-based phosphorus recovery in fertiliser production.

Several limitations merit consideration in interpreting these findings. The Environmental Life Cycle Assessment (E-LCA) employs a cradle-to-gate boundary and uses secondary data sources (e.g., generic database processes for energy), which may not fully reflect site-specific operational conditions or regional supply chain variations. The social evaluation, scoped as an upstream SHA leveraging the Social Hotspots Database (SHDB), does not imply plant-level inventory data for fertiliser production and end-use phases; a comprehensive Social Life Cycle Assessment (S-LCA, encompassing SLCPA and SLCIA) would necessitate detailed primary data from individual facilities. Moreover, the P recovery process modelling draws from pilot-scale data with conservative assumptions regarding chemical inputs and energy demands, likely overstating impacts compared to prospective industrial implementations. Finally, agronomic performance and long-term soil accumulation effects from substituting mineral phosphorus with SSA-derived alternatives fall outside this study's scope and need to be object future investigation.

To enhance CE solutions employment, refinements to the wet acid leaching stage should be prioritised - such as optimised reagent selection, dosage control, and recycling - alongside reductions in energy intensity and water consumption within the recovery pathway, consistent with insights from recent LCA evaluations of SSA-based and hydrothermal phosphorus recovery routes. The observed advantages of LG-MgO over $Ca(OH)_2$ affirm its potential as a pivotal lever for curbing global warming potential and fossil resource depletion, corroborating findings from magnesium-centric phosphorus recovery literature. Social assessment advancement hinges on stakeholder collaboration (including mining operators, fertiliser manufacturers, and local communities) to gather facility-specific social inventories, facilitating progression from screening-level SHA to robust cradle-to-gate S-LCA inclusive of production and, where practicable, application phases. Process synergies between phosphorus recovery units and incumbent fertiliser facilities - via heat recovery, shared infrastructure, and on-site neutralisation - promise further environmental relief and economic viability. Furthermore, future studies should extend the system boundaries to include the use phase and investigate potential differences in fertiliser efficiency and nutrient bioavailability between BAU and CE formulations.

Finally, it should be remarked that the environmental and social results presented here are representative of the specific, well-characterised SSA investigated within the PHOSTER project. While the recovered P-based products obtained under the analysed configurations complied with EU regulatory limits for fertilising products, SSA composition and heavy metal content may vary depending on wastewater treatment configurations, sludge origin and incineration conditions. Therefore, future research should assess the robustness of recovery performance and regulatory compliance across a wider range of SSA typologies.

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CHAPTER 7

Life Cycle Assessment and Circularity evaluation of sustainable olive pomace valorization through innovative High-Pressure High-Temperature Extraction

This chapter is extensively based on the following publication, under review as of January 2026:

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7 LIFE CYCLE ASSESSMENT AND CIRCULARITY EVALUATION OF SUSTAINABLE OLIVE POMACE VALORIZATION THROUGH INNOVATIVE HIGH-PRESSURE HIGH-TEMPERATURE EXTRACTION

ABSTRACT

Olive pomace (OP), a major by-product of olive oil production, represents both an environmental burden and a resource opportunity. This study performs a prospective Life Cycle Assessment (LCA) combined with circularity metrics to evaluate the environmental performance, resource recovery potential, and circular efficiency of producing value-added biopesticides from OP using an innovative High-Pressure High-Temperature Extraction (HPHTE) process, compared to conventional extraction. This study implies a prospective LCA approach, which is particularly relevant as it enables the anticipation of environmental impacts and improvement potentials of emerging technologies before their industrial implementation, supporting informed decision-making in early design stages. Consequently, experimental data from laboratory-scale trials were upscaled to simulate pilot-scale operation, thus enabling the prospective assessment of emerging extraction technologies. Two biopesticide formulations were analyzed: a freeze-dried powder (Biopesticide A) and a refrigerated liquid (Biopesticide B). Results show that HPHTE consistently reduces energy demand, greenhouse gas (GHG) emissions, and water-related impacts, while increasing polyphenol yields compared to conventional extraction. Circularity metrics reveal that Biopesticide B attains higher material circularity, although with less favorable environmental performance. The prospective scaling-up of HPHTE further enhances environmental benefits and process efficiency. Overall, the integrated assessment demonstrates that valorizing OP through HPHTE provides a sustainable pathway for transforming agro-industrial residues into high-value bioproducts, contributing to circular economy strategies and offering early-stage guidance for technology development and upscaling.

7.1 INTRODUCTION

Systemic approaches that together manage water, energy, food and climate interdependencies throughout agri-food chains are necessary to meet the 2030 Agenda. In this context, the principles of the circular economy (CE) can support SDGs 9 (industry, innovation, and infrastructure) and 12 (responsible consumption and production) while lowering resource consumption and environmental burdens (Fund, 2015). The global shift towards sustainability has increased interest in CE principles (Geissdoerfer et al., 2017). The CE emphasizes strategies such as the reduction, reuse and recycling of materials throughout the entire production cycle (Kirchherr et al., 2023). Products and processes are designed to prioritize longevity, while by-products and waste streams are reimagined as valuable resources rather than simply waste (Rifna et al., 2023). In the agri-food sector vast amounts of waste remain underutilized making agri-food waste a pressing challenge by wasting valuable resources and also significantly contributing to greenhouse gas (GHG) emissions and climate change (Demichelis et al., 2025). According to data from the Food and Agriculture Organization (FAO), about one-third of the food produced globally in 2013 was wasted (FAO, 2013).

On average, the EU produces around 2 million tons of olive oil annually, with Spain leading production (66%), followed by Italy, the second-largest producer (15%), and Greece (13%) (https://worldpopulationreview.com/country-rankings/olive-oil-production-by-country). However, the olive oil industry lead to large quantities of waste and by-products owing high organic and phenolic contents, making difficult their management and disposal (Cinardi et al., 2024). The recycling of the olive oil wastes and by-products in olive orchards management can potentially be beneficial to sustainability, but presents risks of environmental pollution (Zipori et al., 2020). Conversely, valorizing olive oil by-products presents significant commercial potential, particularly for the recovery of antioxidants and fatty acids that are valuable for human nutrition (Khdair and Abu-Rumman, 2020; Rodríguez-Pérez et al., 2025). In this regard, olive pomace (OP), with an estimated global production of ~2 million tons per year, treated as waste poses disposal challenges due to its high organic content and potential environmental risks (Enaime et al., 2024). Consequently, transitioning OP from waste to resource aligns with the principles of the CE (Scandurra et al., 2023). OP valorization is enhanced by its composition, since it contains phenolic compounds, dietary fibers, and bio-lipids that that can be recovered to obtain added-value products such as antioxidants and bioactives (Duman et al., 2020; Madureira et al., 2022; Osorio et al., 2021). Valorization of OP is feasible via different methods. At the laboratory scale, numerous studies have investigated innovative extraction methods aimed at recovering phenolic compounds and other bioactives with antioxidant, antimicrobial, and biopesticide properties (Tapia-Quirós et al., 2022).

Techniques such as high-pressure high-temperature extraction (HPHTE), ultrasound-assisted extraction, and supercritical CO_2 extraction have been explored for their efficiency and selectivity (Casazza et al., 2012; Stramarkou et al., 2023; Tapia-Quirós et al., 2022). However, while lab-scale results demonstrate promising yields and product quality, scaling these processes to pilot and full-scale remains challenging due to energy requirements, economic feasibility, and process integration within existing olive oil supply chains (Ribeiro et al., 2020; Rodríguez et al., 2022).

Indeed, at the industrial level, full-scale OP valorization approaches have primarily focused on energy recovery through anaerobic digestion, combustion, or composting, which reduce disposal impacts but often fail to maximize the recovery of high-value compounds (Demichelis et al., 2025; Gómez-Cruz et al., 2024; Ochando-Pulido et al., 2024). Most studies have mainly focused on energetic valorisation, instead of the production of added-value products (García Martín et al., 2020; Khounani et al., 2021). Bridging the gap between laboratory-scale innovation and full-scale industrial deployment is therefore essential to achieve both environmental sustainability and economic viability. Life Cycle Assessment (LCA), performed according to standards ISO 14040-044 plays a pivotal role in narrowing this gap by providing a systematic framework to evaluate the environmental implications of emerging processes across their entire value chain (Del Borghi et al., 2020; ISO, 2006a, 2006b). By quantifying resource use, emissions, and potential impacts from laboratory to full-scale deployment, LCA enables the identification of environmental hotspots and informs technology optimization before large-scale implementation (Gallo et al., 2022; Moreschi et al., 2024). In this context, the use of a prospective LCA approach allows to anticipate the potential environmental performance of emerging technologies prior to industrial implementation, supporting early-stage design and decision-making (Arvidsson et al., 2018; Thonemann et al., 2020).

Despite a growing interest of CE strategies implementation in the agri-food sector, limited studies have systematically assessed environmental impacts of innovative processes compared to conventional ones employed for waste valorization (Mondello et al., 2024). In addition, the combination of LCA and CE indicators in this field is still limited (Romero-Perdomo and González-Curbelo, 2023).

The present study aims at performing an integrated prospective assessment of HPHTE process for the valorization of OP into added-value products, such as biopesticides, within CE framework. To achieve this goal, environmental impacts were quantified through LCA and complemented by circularity metrics. Specifically, a prospective LCA approach was adopted to anticipate the environmental performance of HPHTE prior to its industrial implementation, thus supporting early-stage decision-making and design improvements

Innovative HPHTE was compared with solvent-based conventional extraction in terms of total polyphenol yield and environmental impacts to provide additional evidence on processes sustainability and effectiveness and to anticipate their environmental performance under pilot-scale deployment conditions. Laboratory-scale data were then upscaled to simulate pilot-scale operations for both HPHTE and conventional extraction, enabling a prospective evaluation of environmental trade-offs and synergies, and ensuring that technological scaling aligns with sustainability objectives.

7.2 MATERIALS AND METHODS

7.2.1 Experimental study of polyphenol extraction from OP

Olive pomace (OP) of Taggiasca cultivar (Olea Europea L.) was collected from a three-phase oil extraction decanter operated at an olive oil industry located in Liguria region (Italy).

As shown in **Figure 7.1**, the valorization process encompasses successive steps aimed at recovering polyphenolic and producing finished products, biopesticides in two different formulations. In the drying phase, OP was oven-dried at 60 °C for approximately 48 h until a constant moisture content (5-8%) was reached (Aliakbarian et al., 2011). After drying, olive pomace (30.0 ± 0.1 g) was subjected to extraction by HPHTE using an ethanol-water solution (1:1, v/v) at a solid-to-liquid ratio of 1:10. Extractions were carried out in an HPHTE reactor (Model 4842, Parr Instrument Company, Moline, IL, USA). Nitrogen flushed into the reactor at a flow rate of 1 L/min for 1 min to establish an inert atmosphere and prevent phenolic oxidation. The HPHTE reactor was hermetically sealed, ensuring that internal pressure was directly proportional to temperature. Extractions were performed under established conditions (180 °C, 15 bar, and 90 min) as reported previously (Aliakbarian et al., 2011; Paini et al., 2016). For comparison, conventional extraction (labelled as CONV) was carried out using an ethanol-water solution (1:1, v/v) at a solid-to-liquid ratio of 1:10. The mixture was stirred on a magnetic plate (LLG-uniSTIRRER 3) for 24 hours at room temperature (25 ± 1 °C).

Afterwards, the obtained extracts (from HPHTE or CONV) were centrifugated at 6000 x g for 10 minutes (centrifuge model MF 20, Alliance Bio Expertise, Guipry, France) and subsequently, the extracts were filtered under vacuum using a Büchner funnel, yielding a purified extract and a wet residue. Total polyphenol content was quantified on purified extract according to (Duman et al., 2020), using caffeic acid as the reference standard, and consequently expressed as milligrams of caffeic acid equivalents per g of dried pomace (mgCAE/gDP).

The extract was then concentrated by rotary evaporation (Heidolph Laborota 4000), after which ethanol was replaced with an equivalent volume of water.

Lastly, two formulations were obtained: (i) a powdered biopesticide (Biopesticide A), produced by freeze-drying process (model D-37520, CHRIST Freeze-Dryers, Osterode am Harz, Germany), and (ii) a liquid biopesticide (Biopesticide B), obtained after evaporation and stored in a refrigerator (at 4 °C) for up to one month (**Figure 7.1**).

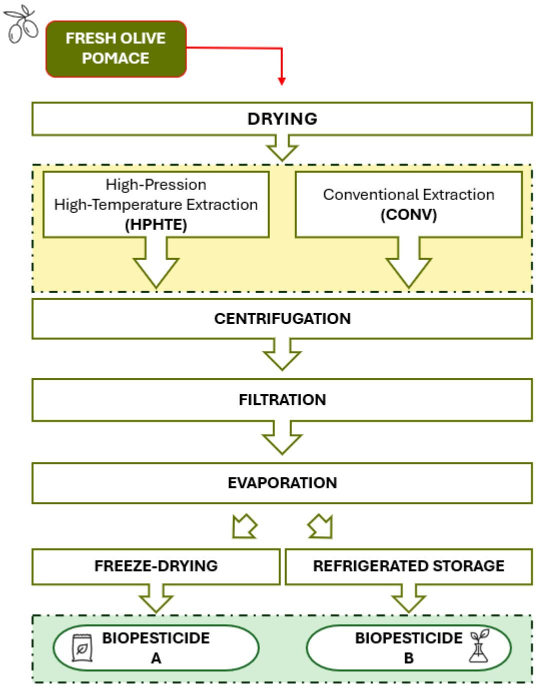

Figure 7-1: Schematic view of lab-scale experimental study.

7.2.2 Life Cycle Assessment Methodology and scenarios description

One quantitative technique that might support the application of a life cycle thinking approach is LCA methodology. An LCA makes it easier to conduct a systematic quantitative evaluation of goods and services in terms of environmental impacts, resource consumption and human health. The LCA study reported in this work was carried out in compliance with ISO 14040 and ISO 14044 international standards (ISO, 2006a, 2006b) by carrying out four phases: Goal and Scope Definition, Inventory Analysis, Impact Assessment, and Interpretation. The specification of the LCA goal and scope gives a description of the product system being studied, including its functional unit and system boundaries.

The overall objective of the LCA study is the valorization of olive pomace, considered as a waste material, through a process that allows its transformation into added value and marketable products. The software used for LCA modelling is SimaPro 10.2. Foreground data, relating directly to the product system under study, were obtained from primary data collection while background data were sourced from Ecoinvent 3.11 database (Frischknecht and Rebitzer, 2005; Wernet et al., 2016).

The main function of the investigated system is OP treatment, consequently as the Functional Unit (FU) chosen in this study is the management of 1 kg of fresh OP. The choice 1 kg as the FU for waste management processes is typical for waste management studies (Elginoz et al., 2020) and recommended when the pre-disposal life cycle stages of the products generating the waste are not included (Del Borghi et al., 2009).

To assess the environmental performance of HPHTE as innovative technique for olive pomace valorisation and the production of bioproducts three different scenarios were investigated. The first scenario compared two products owing different formulations and both derived from HPHTE technique. The first one labelled as “Biopesticide A” is a freeze-dried powder that offers chemical stability, a long shelf life, and simplicity of transportation, while the other labelled as “Biopesticide B” is a liquid that may be kept in an aqueous solution under refrigeration for up to one month, making application simpler and faster but needed accurate transport and storage. Biopesticide A's lyophilization phase and Biopesticide B's refrigeration step result in distinct electrical demands, the comparison analysis took stability, storage, usefulness, environmental impact, and energy consumption into account. Fresh olive pomace is gathered at the mill and driven to the lab, where it is subjected to residual oil extraction, drying, HPHT extraction, centrifugation, filtration, and evaporation. The two biopesticides and a dephenolized wet residue are the three material outputs of the entire process.

The second scenario compared HPHTE and conventional extraction (labelled as CONV), while other steps are the same. Conventional extraction is generally associated with lower selectivity and potentially higher energy consumption, as reported in previous studies (Casazza et al., 2012; Selvamuthukumaran and Shi, 2017). Conversely, HPHTE is an innovative method that allows for optimum operating conditions to maximize polyphenolics yield and to decrease waste.

For third scenario, processes employed at lab-scale and described in the previous section were scaled up according to methodological approach proposed by Piccino et al. (2016). Consequently, the scale-up framework with an LCA perspective was performed to obtain logically and systematically organized data, enabling simulation of the process at a larger scale (Piccinno et al., 2016). **Table 7.1** summarized the relevant assumptions made for the scaling-up. This scenario demonstrates how local by-product valorization can lead to increased energy efficiency and improved circularity.

Table 7-1: Summary of the relevant assumptions of second (lab-scale conventional extraction) and third scenario (scale up to 20 liters extractor) with respect to the first (HPHTE lab-scale, baseline) scenario.

Aspect	**Second Scenario (Lab-scale setup)**	**Third Scenario (Scaling-up)**
Functional unit	1 kg of fresh OP treated	1 kg of fresh OP treated
Objective	Comparison between HPHTE and conventional extraction	Evaluation of scale-up efficiency and environmental improvement through local valorization
Amount of treated olive pomace (OP)	30.0 ± 0.1 g dry-mass per batch (normalized to 1 kg of OP for inventory)	1.5 kg dry-mass per batch in 20 L HPHT reactor (normalized to 1 kg of OP for inventory)
Plant location	Laboratory of University (Liguria, Italy)	Pilot-scale plant located at or near the olive mill
Transport phase	OP transported from olive mill to laboratory (118 km measured distance)	Negligible transport (co-located system at olive mill)
Feedstock	Fresh Taggiasca olive pomace from three-phase decanter	Same feedstock (Taggiasca OP) directly used on-site
Drying	Oven-drying at 60 °C for 48 h	Industrial hot-air dryer
Conventional Extraction	Conventional extraction, suspension was magnetically stirred at room temperature (25 ± 1 °C) for 24 hours using an LLG-uniSTIRRER 3.	Conventional extraction was replicated using the same solvent ratio (ethanol-water 1:2, v/v) and solid-to-liquid ratio (1:10) as in the laboratory setup. The process was assumed to be performed in a 20 L stirred reactor at ambient temperature (25 ± 1 °C) for 24 h, with a total specific electricity consumption of 94 kWh kg^{-1} of fresh olive pomace treated.
HPHTE	Nitrogen flushed into the reactor at a flow rate of 1 L/min for 1 min. The reactor was hermetically sealed, ensuring that internal pressure was directly proportional to temperature. at 180 °C, 15 bar, and 90 min	Pilot HPHT extractor (20 L total, 15 L useful); heating 6 kW + agitation 0.1 kW for 90 min

Aspect	**Second Scenario (Lab-scale setup)**	**Third Scenario (Scaling-up)**
Solvent composition	Ethanol-water (1:1 v/v), solid-to-liquid ratio 1:10	Same solvent ratio: Ethanol-water (1:1 v/v), solid-to-liquid ratio 1:10
Centrifugation	6000 × g, 10 min	Industrial centrifuge, similar energy per kg OP
Filtration	Büchner funnel vacuum filtration	Industrial vacuum filtration
Evaporation	Rotary evaporator	Pilot rotary evaporator with improved heat recovery
Freeze-drying	Laboratory freeze-dryer	Industrial freeze-dryer process
Refrigerated storage	Laboratory refrigerator	Cold room storage
Final products	Biopesticide A (freeze-dried), Biopesticide B (liquid), and dephenolized wet residue	Same outputs; larger scale enables on-site valorization and potential reuse of residue

For this study, all scenarios' system boundaries were analyzed using a "cradle-to-gate" approach, including the steps needed for OP treatment and the subsequent processes of valorization to produce biopesticides.

Iterative data gathering and data compilation in a Life Cycle Inventory (LCI) table are part of the Inventory analysis phase of the Life Cycle Assessment (LCA). The most labor-intensive and time-consuming part of life cycle assessments (LCAs) is frequently identifying the inputs and outputs for a particular product system across its life cycle (Rebitzer et al., 2004). The inventories for each scenario are presented in **Table 7.2**. Each table reports the input and output quantities referred to the specific phase of the valorization process. Data regarding the valorisation are directly collected from laboratory processes, including material and energy inputs related to the HPHT extraction. The same applies for pilot scale 20 liters extraction. Instead, for the conventional extraction no primary data were available and thus secondary data from the Ecoinvent 3.11 database (Wernet et al., 2016) were used both for the laboratory scale and for the pilot scale. All data are referred to the functional unit of 1kg of fresh olive pomace. For the sake of completeness, complete LCI for lab-scale and simulated pilot-scale processes is reported in **Table D-1** (Appendix D).

Table 7-2: Life Cycle Inventory for the baseline (HPHTE at lab-scale), for 1 kg of fresh olive pomace (FU).

Phase	Category	U.M.	Quantity
DRYING			
	Energy consumption	kWh	56.26
HPHTE			
	Water	kg	3.15
	Ethanol	kg	2.50
	Nitrogen	l	21.00
	Energy consumption	kWh	11.11
CONV EXTRACTION			
	Water	kg	3.15
	Ethanol	kg	2.50
	Energy consumption	kWh	259.55
CENTRIFUGATION			
	Energy consumption	kWh	0.97
FILTRATION			
	Energy consumption	kWh	0.50
EVAPORATION (for the extract)			
	Tap water	l	839.98
	Energy consumption	kW	29.40
	Wastewater	l	839.98
FREEZE-DRYING (to obtain Biopesticide A)			
	Energy consumption	kWh	282.23
REFRIGERATED STORAGE (to obtain Biopesticide B)			
	Energy consumption	kWh	24.25
Waste disposal			
	Waste (wet residue)	kg	0.94

The Life Cycle Impact Assessment (LCIA) determines and assesses the possible environmental effects of the LCI, including their magnitude and importance. The environmental impacts of the two systems under analysis (HPHTE and CONV) at laboratory scale and pilot scale are measured through the Product Environmental Footprint 3.1 (PEF 3.1) method (Joint Research Centre (European Commission) et al., 2023). The Cumulative Energy Demand

(CED) method (Frischknecht et al., 2007) was used to examine how much energy was used during the different processes' life cycle. This covers both the direct and indirect, or gray, energy consumption brought on by the use of materials. The climate dimension was assessed through the Global Warming Potential (GWP) indicator, following IPCC characterization factors (IPCC, 2021). **Table 7.3** reports the relevant impact categories selected, related acronyms, units of measure and references.

Table 7-3: Summary of the relevant impact categories selected.

Impact category	Acronym	U.M.	Reference
Global Warming Potential	GWP	Kg CO_2 eq.	(IPCC, 2021)
Cumulative Energy Demand	CED	MJ	(Frischknecht et al., 2007)
Acidification potential	AP	mol H+ eq	(Seppälä et al., 2006)
Eutrophication, marine	EuM	kg N eq	(Goedkoop et al., 2009)
Eutrophication, freshwater	EuF	kg P eq	(Goedkoop et al., 2009)
Eutrophication, terrestrial	EuT	mol N eq	(Seppälä et al., 2006)
Water use	WU	m3 eq.	(Boulay et al., 2018)

7.2.3 Circularity Indicators

To complement LCA findings with a measure of resource circularity, two product-level indicators were calculated: the Material Circularity Indicator (MCI) (EMF, Granta Design, 2015) and the Material Reutilization Score (MRS). With an emphasis on the recovery and value-adding of material flows, these metrics were created to evaluate how closely processes or products adhere to the CE principles.

The MRS measures a product's potential for recycling. MRS combines recycled content (RC, percentage of recycled material utilized) with intrinsic recyclability (IR, percentage of product recyclable at least once). A score ranging from 0 to 100% is produced by weighting IR twice as much as RC (Niero and Kalbar, 2019). Recyclability ratings for olive pomace residues and auxiliary inputs like water and ethanol were assigned in this study based on the composition of the biopesticide production process streams.

The Ellen MacArthur Foundation and Granta Design created the MCI, which assesses a product system's restorativeness in relation to material fluxes. It has a range of 0 (completely linear) to 100 (completely circular). Quantifying the percentage of recycled content, the rates of collection and reuse, and the effectiveness of recycling procedures are all necessary steps for the evaluation of this indicator (Niero and Kalbar, 2019). The flows of olive pomace entering the drying phase and high-pressure high-temperature extraction processes were used to compute the MCI for the case studies. The percentage of circular inflows (i.e., recycled or bio-based content) and circular outflows (recoverable residues) were compared after the mass balance of inputs (olive pomace, water, and ethanol) and outputs (biopesticide product, residues) were reconstructed.

7.3 RESULTS AND DISCUSSION

7.3.1 Life Cycle Assessment

The results are presented sequentially: first, the laboratory-scale LCA is analyzed by unit operations; second, HPHTE is compared with conventional extraction; third, the impact of product formulation is assessed; and finally, the effects of scaling up to a 20-L pilot system are discussed.

The life cycle inventory for the HPHTE process was elaborated at laboratory scale and normalized to 1 kg of treated olive pomace (OP), whereas **Table 7.4** summarizes the environmental profile of the HPHTE baseline process.

Table 7-4: LCA impact categories results for the baseline scenario (HPHTE at laboratory scale).

Impact Category [U.M.]	Transport	Drying	HPHTE	Centrifugation	Filtration
GWP [kg CO2 eq]	45.60	19.80	10.11	0.34	0.21
AP [mol H+ eq]	0.12	0.08	0.08	0.00	0.00
EuM [kg N eq]	0.02	0.01	0.07	0.00	0.00
EuFW [kg P eq]	0.01	0.01	0.00	0.00	0.00
EuT [mol N eq]	0.24	0.14	0.31	0.00	0.00
CED [MJ]	600.11	466.66	407.28	8.01	4.68
WU [m3 depriv.]	3.54	10.05	3.26	0.17	0.10

Impact Category [U.M.]	Evaporation	Freeze drying	Refrigerated storage	Residual waste treatment
GWP [kg CO_2 eq]	11.02	99.27	8.53	0.01
AP [mol H+ eq]	0.05	0.40	0.03	0.00
EuM [kg N eq]	0.02	0.06	0.01	0.00
EuFW [kg P eq]	0.00	0.03	0.00	0.00
EuT [mol N eq]	0.09	0.69	0.06	0.00
CED [MJ]	254.17	2340.17	201.07	0.15
WU [m3 depriv.]	9.30	50.39	119.68	0.01

Among all impact categories, freeze-drying and transport stages emerged as the most energy- and emission-intensive steps. Freeze-drying accounted for approximately 70-75% of the total Cumulative Energy Demand (CED) and Global Warming Potential (GWP), due to the prolonged operation time and high thermal load associated with sublimation under vacuum. The transport phase also significantly contributed to GWP (45.6 kg CO_2 eq), highlighting the influence of logistics between the olive mill and laboratory site.

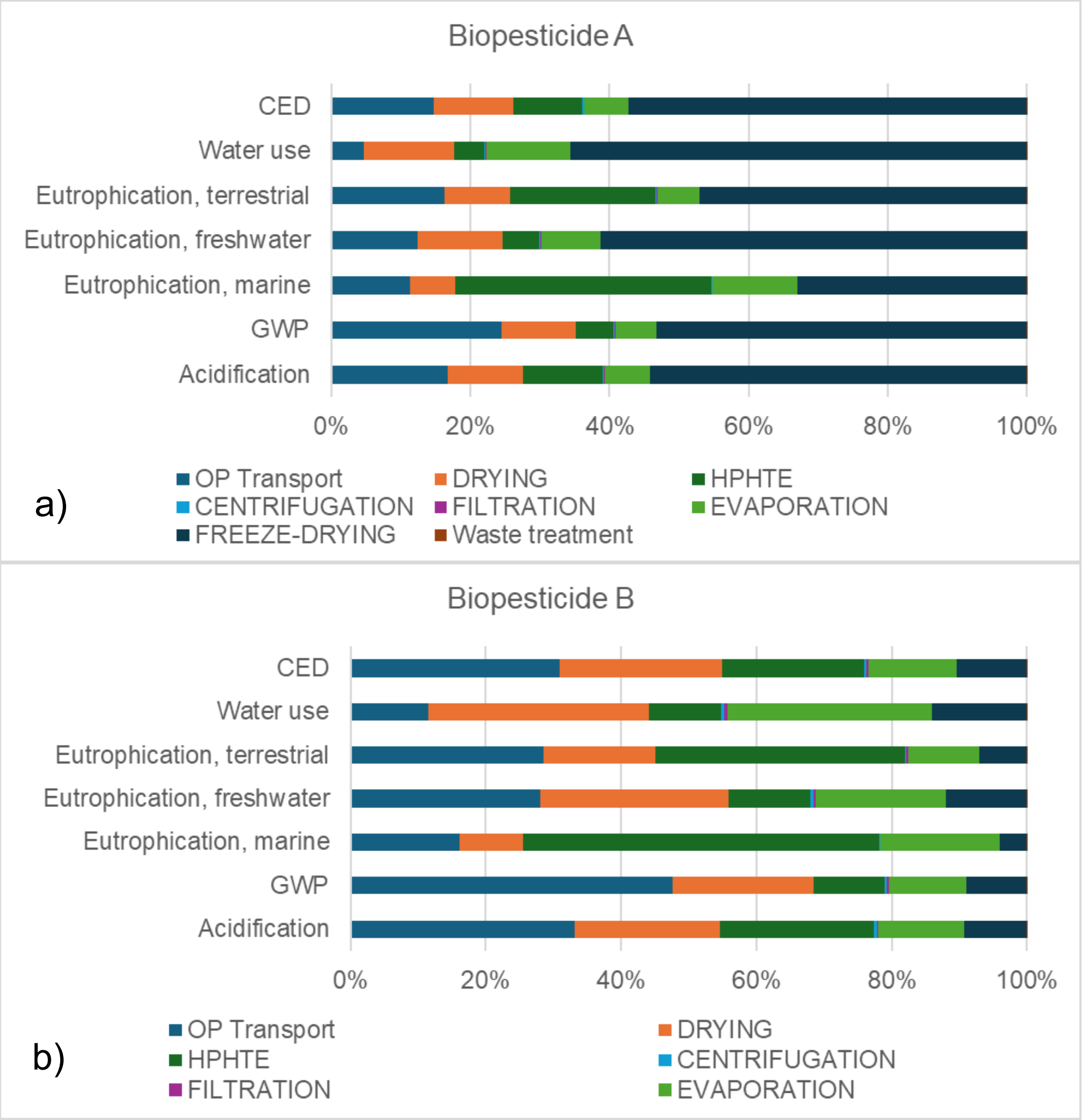

Figure 7-2: Relative contribution of the single phases to the production of Biopesticide A (a) and Biopesticide B (b).

The HPHTE extraction represented a lower share of the total environmental burdens (10.11 kg CO_2 eq, 407.28 MJ), whereas drying and evaporation were secondary contributors, mainly due to electricity consumption for heating. Centrifugation, filtration, and residual waste treatment were found to have negligible impacts across all categories (<2%). Overall, the process displayed the highest relative contributions in the categories of Global Warming Potential (GWP = 194 kg CO_2 eq), Cumulative Energy Demand (CED = 4300 MJ), and Water Use (WU = 196 m^3 depriv.), confirming that the main environmental hotspots are energy-related operations and solvent evaporation stages.

Tables D2-D5 (Appendix D) report the complete dataset of LCIA results for all impact categories and unit operations related to the production of Biopesticide A and B at laboratory scale through HPHTE and Conventional extraction (CONV).

It is interesting to notice that, being the processes the same for the production of both biopesticides with the exception of the final step, the impact of freeze-drying is much higher than the impact of refrigerated storage (**Figure 7.2**). In terms of relative contribution to the total impact, freeze-drying process accounts for ~50% across all the impact categories explored, while refrigerated storage less than 20%.

Figure 7.3 compares the environmental performance of the two final products obtained through HPHTE Biopesticide A (freeze-dried) and Biopesticide B (liquid refrigerated). Although both share identical upstream processes (drying, HPHTE, centrifugation, filtration, evaporation), their final treatment steps differ substantially in energy consumption.

Freeze-drying (Biopesticide A) showed the highest environmental burden, influencing all impact categories. In contrast, Biopesticide B required less than half of the total energy, resulting in a 49-60% reduction in GWP, CED, and AP. However, the liquid formulation has shorter shelf life and requires refrigerated storage, which still contributes to overall energy demand.

Therefore, the selection between the two formulations involves a trade-off between stability and energy intensity. Biopesticide A ensures long-term preservation and easier transport. Biopesticide B minimizes environmental impacts but entails limitations in logistics and shelf-life.

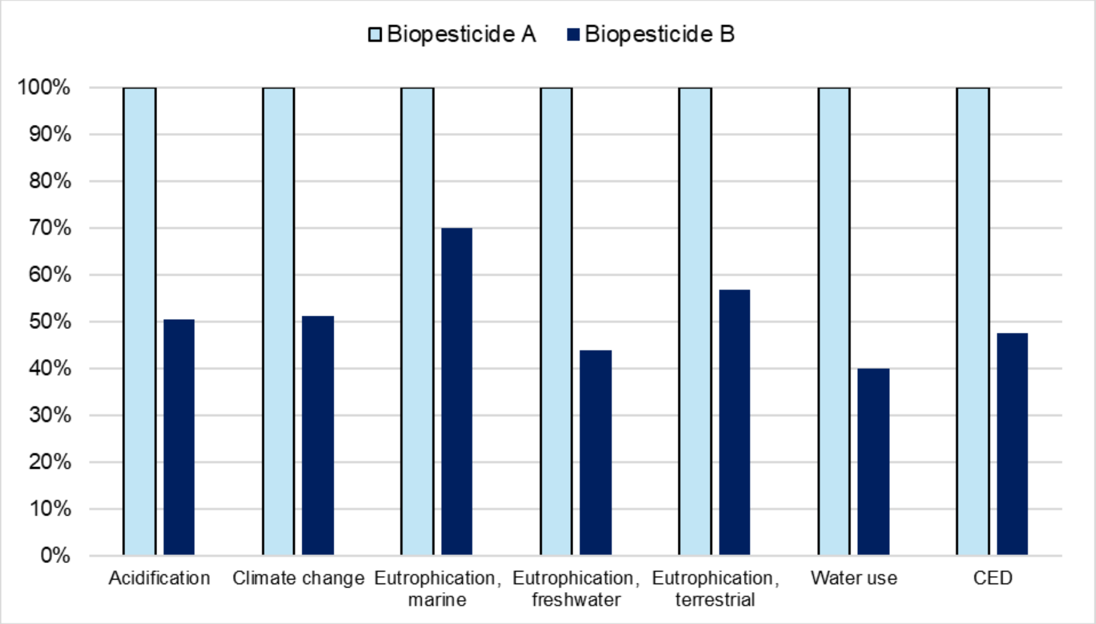

Figure 7-3: Comparison of Biopesticide A and Biopesticide B at laboratory scale.

When compared to the conventional ethanol-water extraction (CONV), the HPHTE process demonstrated a consistent reduction in environmental burdens across all impact categories (**Figure 7.4**).

At laboratory scale, HPHTE showed reductions of 83% in GWP, 83% in CED, and 93% in water use compared to the conventional extraction. Significant improvements were also observed in acidification (-81%) and terrestrial eutrophication (-66%). These results indicate that the high-pressure, high-temperature process enhances extraction efficiency, reducing solvent requirements and extraction time while achieving higher polyphenolic yields. The environmental benefits of HPHTE over CONV are primarily associated with the shorter extraction duration (90 min vs. 24 h) and the absence of long magnetic stirring cycles, which drastically lower electricity demand. Despite the higher energy intensity per unit of time, the reduced process duration leads to a lower total energy requirement per kg of OP processed.

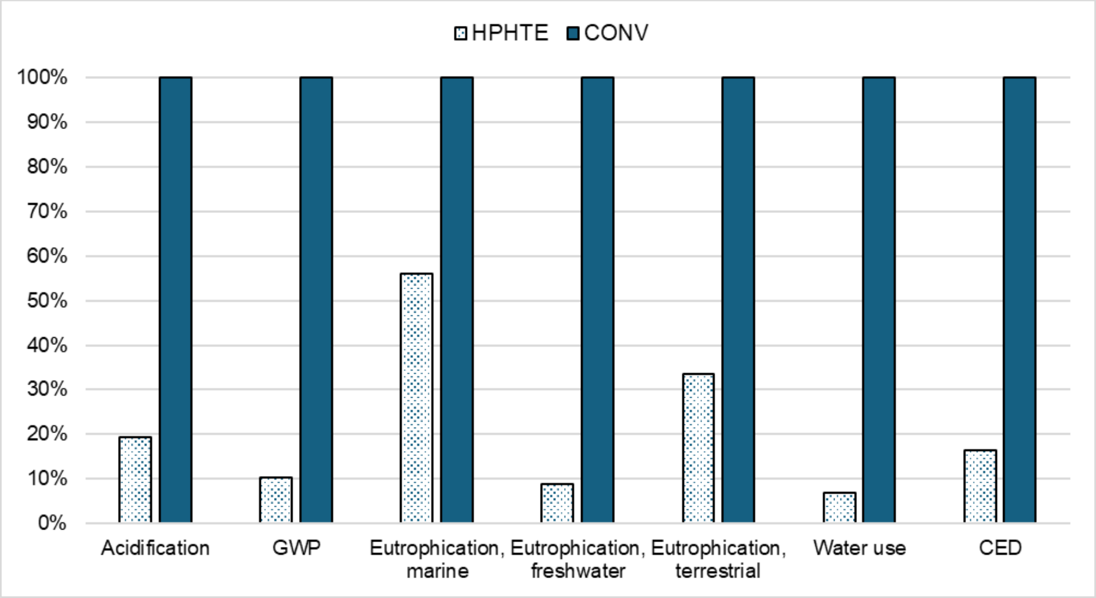

Figure 7-4: Comparison of HPHTE and CONV at lab-scale (Second Scenario).

The yield of total phenolic compounds obtained from HPHTE and CONV varied significantly. The measured values were 63.13 mgCAE/gDP for HPHTE and 5.12 mgCAE/gDP for CONV. This demonstrates that the same amount of input can yield higher-value bioproducts, promoting resource efficiency and a circular bioeconomy. HPHTE had a substantially lower GWP, demonstrating that enhanced yields and energy-efficient operations lower emissions per functional unit. HPHTE performs around 12 times better than CONV and can be considered consequently an interesting opportunity for scaling-up.

The scale-up of the HPHTE process from laboratory to a 20-L pilot-scale was assessed by hypothesizing that the upscaled system, located near the olive mill, eliminates the transportation phase and benefits from higher process efficiency and heat integration. These pilot-scale simulations provide a prospective assessment of HPHTE, highlighting potential environmental benefits before full industrial implementation. When expressed per kg of treated OP, the scaled-up HPHTE process displayed notable environmental improvements compared to both the laboratory HPHTE and the scaled-up conventional extraction.

Figure 7.5 shows that HPHTE, at pilot scale, achieved reductions of approximately -74% GWP, -59% CED, -69% water use, -59% acidification, and -41% terrestrial eutrophication compared to CONV-pilot scale.

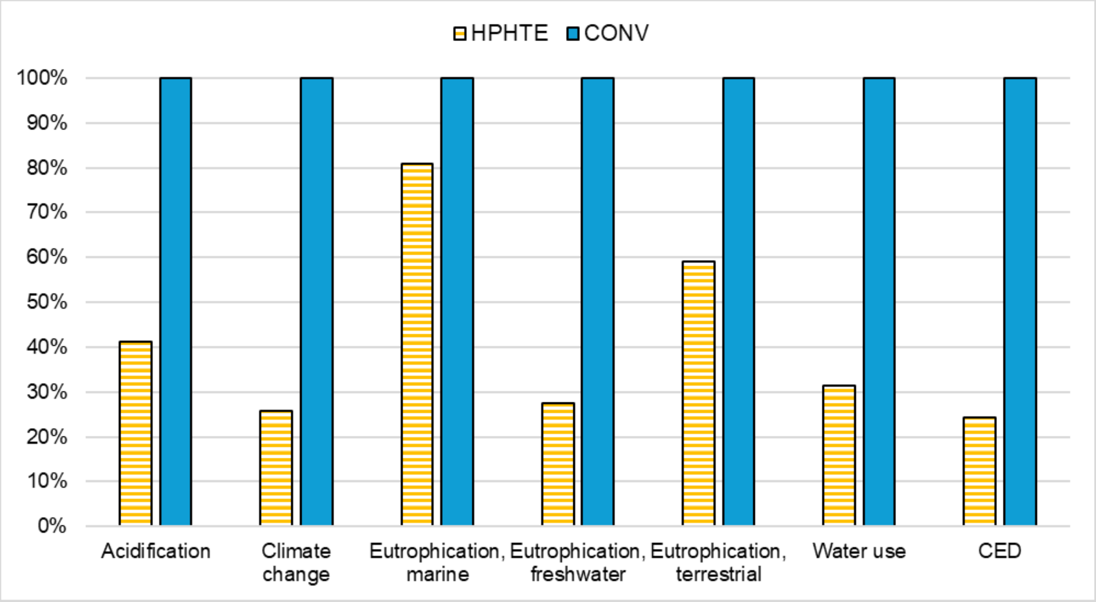

Figure 7-5: Results from scaling-up and comparison between HPHTE and conventional extraction (CONV) (Third Scenario).

Within the scaled-up HPHTE, the main contributors remained the extraction step (≈60% of GWP and CED) and drying (≈10-20%), while other operations had marginal relevance (<5%). In contrast, in the CONV-pilot scale, the extraction phase alone accounted for over 85-90% of all impact categories, confirming its lower energetic and process efficiency (**Figure 7.5**).

While the comparative performance of HPHTE and conventional extraction is robust across the selected impact categories, several parameters may influence the magnitude of the results. The most critical sources of uncertainty relate to the electricity mix, solvent recovery efficiency, and polyphenol yield. In particular, varying the electricity supply from the national to a renewable-enriched mix could reduce the climate impact of both extraction routes by up to 25-30%, with a proportionally higher benefit for the more energy-intensive freeze-drying step. The upscaling of HPHTE from laboratory to pilot scale introduces several uncertainties that can influence the robustness of the environmental results. Key parameters affected by scale-up include energy efficiency, heat recovery, and process integration. Laboratory systems typically operate under less optimized thermal conditions and limited energy recovery, while pilot-scale setups can benefit from process optimization and waste heat valorization, leading to potentially lower specific energy consumption. Conversely, uncertainties related to equipment performance, operational stability, and control of temperature and pressure may counterbalance these gains, especially during the early stages of industrial implementation. Therefore, although absolute values may vary, the relative environmental advantage, within plausible operational ranges, HPHTE consistently maintains a lower impact compared to CONV.

The findings from prospective LCA demonstrate that HPHTE technology maintains its environmental advantage even when scaled up, primarily due to shorter extraction cycles, better solvent-to-solid ratio control, higher polyphenolic yield (reducing solvent losses), and integration of heating and cooling systems. Moreover, the relocation of the pilot-scale unit near the olive mill drastically reduces the transportation burden, contributing to the circularity and local valorization of agricultural residues.

Tables D6-D9 (Appendix D) report the complete dataset of LCIA results for all impact categories and unit operations related to the production of Biopesticide A and B at simulated pilot-scale through HPHTE and Conventional extraction (CONV).

In order to deepen the discussion, the results from simulated pilot-scale HPHTE were analyzed also in terms of final products.

Table 7.5 reports the environmental impacts associated with the HPHTE process performed at simulated pilot scale for both Biopesticide A (spray-dried) and Biopesticide B (refrigerated liquid).

The HPHTE extraction step clearly dominates across all impact categories, particularly in terms of GWP and CED, due to its high energy demand. Downstream processes show distinct contributions: spray drying, required for Biopesticide A, adds significant energy use and emissions, whereas refrigerated storage for Biopesticide B has lower impact. Other operations, including drying, centrifugation, filtration, and evaporation, contribute only marginally. Overall, while both products share the same upstream impacts, the final processing choice determines their environmental profile, with spray drying increasing energy intensity and refrigeration resulting in lower impacts but potentially higher storage constraints.

Table 7-5: LCA impact categories results for HPHTE - pilot scale, process-wise for both final products (Biopesticide A and Biopesticide B).

Impact Category [U.M.]	Drying	HPHTE	Centrifugation	Filtration
GWP [kg CO_2 eq]	0.16	6.60	0.34	0.21
AP [mol H^+ eq]	0.00	0.07	0.00	0.00
EuM [kg N eq]	0.00	0.07	0.00	0.00
EuFW [kg P eq]	0.00	0.00	0.00	0.00
EuT [mol N eq]	0.00	0.28	0.00	0.00
CED [MJ]	3.84	324.42	8.04	4.68
WU [m^3 depriv.]	0.08	1.47	0.17	0.10

Impact Category [U.M.]	Evaporation	Spray dry (Biopesticide A)	Refrigerated storage (Biopesticide B)	Residual waste treatment
GWP [kg CO_2 eq]	1.29	2.72	0.13	0.01
AP [mol H^+ eq]	0.01	0.01	0.00	0.00
EuM [kg N eq]	0.02	0.00	0.00	0.00
EuFW [kg P eq]	0.00	0.00	0.00	0.00
EuT [mol N eq]	0.02	0.02	0.00	0.00
CED [MJ]	25.00	64.01	3.15	0.15
WU [m^3 depriv.]	4.36	1.38	0.07	0.01

Lastly, **Figure 7.6** compares results from scaling-up HPHTE and conventional extraction (CONV) with particular focus on Biopesticide A as the most impactful. The percentage contribution of each process to the selected environmental impact categories are shown.

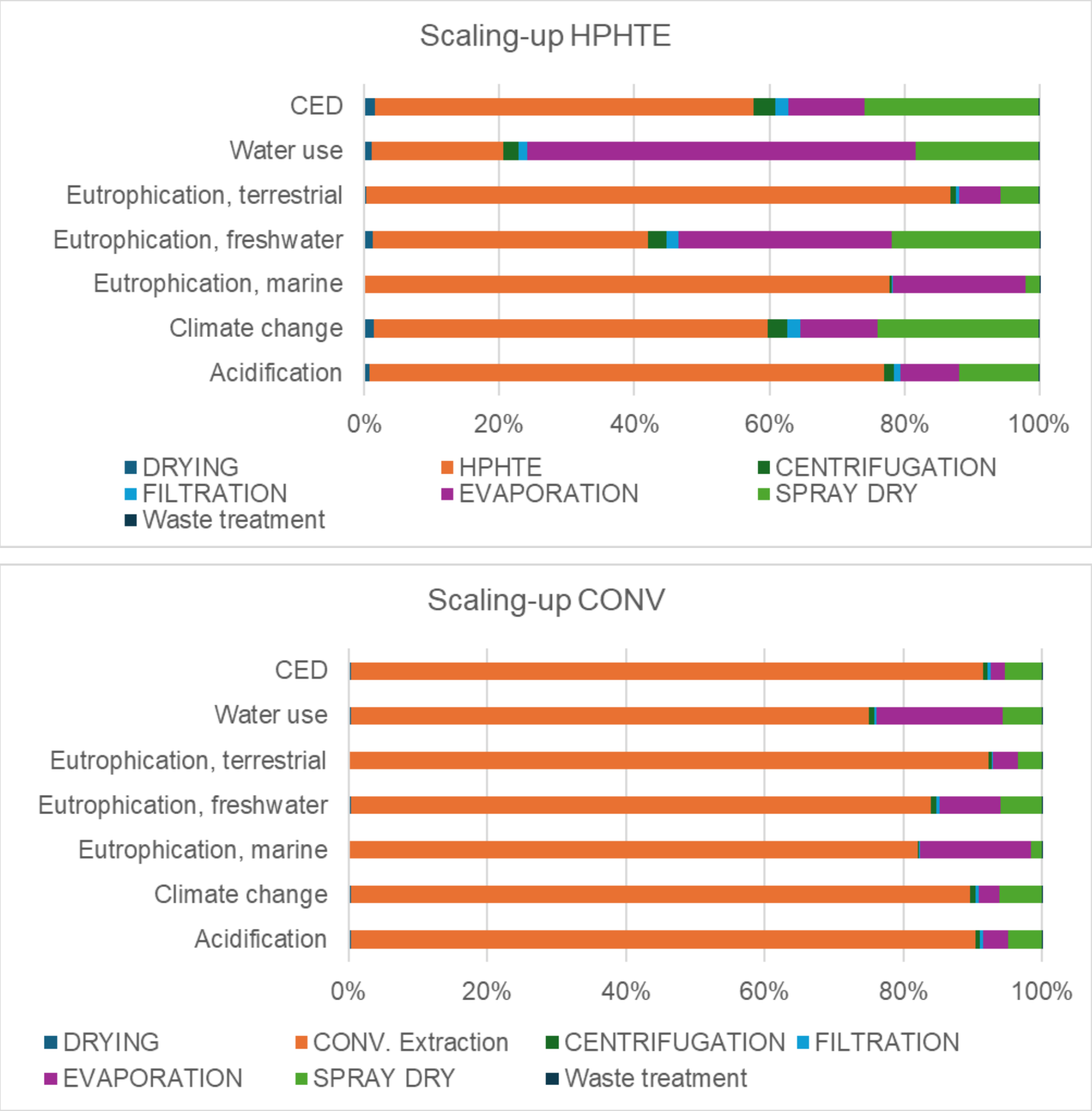

Figure 7-6: Percentage contribution of the single phases to the production of Biopesticide A performing the scaling-up of HPHTE (a) and conventional extraction (CONV) (b).

7.3.2 Circularity Indicators

The Material Circularity Indicator (MCI) and the Material Reutilization Score (MRS), which are shown in **Table 7.6**, were used to measure the circularity performance of the two goods. While Biopesticide B achieved more circularity with an MCI of 36.3% and the same MRS value (3.5%), Biopesticide A achieved an MCI of 5.3% and an MRS of 3.5%.

Table 7-6: Circularity indicators for Biopesticide A and Biopesticide B.

	Material Circularity Indicator MCI	Material Reutilization Score MRS
Biopesticide A	5.3%	3.5%
Biopesticide B	36.3%	3.5%

This study demonstrates that the valorization of olive pomace by HPHTE can improve resource circularity and environmental performance with respect to Conventional Extraction techniques. The circularity indicators, and Integrated Life Cycle Assessment (LCA) enabled thorough examination in line with the Sustainable Development Goals (SDGs) and the concepts of the circular economy (CE).

Overall, the comparative analysis highlights conventional extraction as the least favorable option, while HPHTE-especially under scaled-up conditions-achieves consistently lower burdens across the water, energy, food, and climate dimensions.

CONV had greater levels of eutrophication and freshwater usage, indicating that ineffective extraction techniques could result in increased nutrient releases and wastewater loads. In order to promote water sustainability goals in agro-industrial waste valorization, HPHTE lowers these demands. Despite requiring high-pressure and high-temperature processes, HPHTE's optimized operation uses less energy overall than CONV, proving that advancements in technology can separate energy input from extraction output.

Material Circularity Indicator (MCI) is lower for Biopesticide A (5.3%) than for Biopesticide B (36.3%). Taking into account the finished product, Biopesticide B shows a higher MCI value because it contains water. Water is categorized as a renewable biological material in the MCI framework, increasing the share of renewable inputs. At the same time, this result draws attention to a methodological limitation because the MCI is not able to capture all the aspects of circularity. In addition, the Material Reutilization Score (MRS) was used, and it produces the same result for both products, indicating that the treatment of water as a renewable input-rather than a real improvement in material circularity-is primarily responsible for the apparent difference in MCI.

Biopesticide A's lower Material Circularity Indicator is mainly due to the absence of water in the product. This trade-off between product stability and circularity echoes recent discussions on balancing process energy demands against material recovery outcomes in CE implementation (Falcone et al., 2022).

These valorization pathways directly contribute to SDGs 9 and 12 by fostering innovation, industrial sustainability, and responsible production. The transformation of olive pomace - traditionally an environmental liability - into valuable biopesticides exemplifies circular economy advancements with clear environmental and economic value. However, challenges remain, such as reducing solvent-related impacts, where emerging green solvents and solvent recycling warrant further investigation to enhance process sustainability (Castagna et al., 2025). Thus, HPHTE aligns process efficiency with circular economy objectives by minimizing energy input while maximizing resource retention.

Future research should expand lifecycle boundaries to include end-of-life and application phases of biopesticides for a complete sustainability assessment. Incorporating techno-economic analyses alongside environmental metrics will further inform industrial feasibility, industrial scaling, and policy frameworks supporting the circular bioeconomy of the olive oil industry (Aldana et al., 2023).

7.4 CONCLUSIONS AND FUTURE PERSPECTIVES

This study applied a prospective Life Cycle Assessment (LCA) integrated with circularity metrics to evaluate the environmental performance and resource efficiency of producing value-added biopesticides from olive pomace (OP) using an innovative High-Pressure High-Temperature Extraction (HPHTE) process. Results demonstrated that HPHTE significantly outperforms conventional extraction in terms of energy demand, greenhouse gas emissions, and water-related impacts, while enhancing polyphenol yields and process efficiency.

The prospective nature of the assessment provided critical foresight into the potential benefits and trade-offs of HPHTE during early stages of technological development, highlighting its role as a strategic tool for guiding sustainable innovation. From a circular economy perspective, valorizing OP through HPHTE transforms an abundant agro-industrial residue into high-value bioproducts, closing material loops and promoting more sustainable and resource-efficient bio-based value chains. The integration of environmental and circularity indicators revealed complementary insights. Indeed, while HPHTE enhanced environmental performance, the comparison between biopesticide formulations (liquid vs. powdered) underscores the need for multi-criteria decision-making in process design.

Overall, this study demonstrates that prospective LCA combined with circularity assessment provides a robust framework for steering the sustainable technological development of emerging extraction processes. The HPHTE-based valorization of olive pomace exemplifies a circular bioeconomy pathway capable of reducing environmental burdens, enhancing resource recovery, and fostering innovation toward a more sustainable agri-food sector.

Future research should focus on pilot- and industrial-scale validation of HPHTE systems, integrating process optimization, renewable energy inputs, and solvent recovery strategies to further reduce impacts. Expanding the LCA framework to include techno-economic and social assessments will enable a more holistic sustainability evaluation. Moreover, coupling dynamic and consequential LCA approaches with scenario modeling could capture the long-term implications of technology deployment and market uptake. Consequently, LCA serves not only as a diagnostic tool but also as a cornerstone of eco-design, guiding innovation towards scalable, environmentally responsible valorization pathways. Future integration of process modeling, LCA, and circularity assessment will enable data-driven decision-making for sustainable valorization of agro-industrial residues.

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CHAPTER 8

CONCLUSIONS

8 CONCLUSIONS

This thesis demonstrates that Carbon Farming and circular economy practices can effectively help in tackling the climate neutrality and sustainable development issues. Unlike previous studies, which focused mainly on one dimension, this thesis linked environmental impact assessment (through the use of the LCA methodology), climate mitigation potential quantification (through soil carbon modelling), social hotspot analysis and circularity evaluation. The findings challenge the implementation of Carbon Farming and circular economy practices to be automatically “environmentally friendly” and support the correct design, context evaluation and trade-offs needs to be considered for a fair and scientifically based evaluation.

The five studies presented in this thesis are not isolated investigations but follow a consequential logic that addresses the complexity of agri-food sustainability. The systematic review (**Chapter 3**) initially identified the lack of standardization as a barrier to scaling Carbon Farming, necessitating the spatially explicit modelling approach applied in Flanders (**Chapter 4**). The results from Flanders, identifying the strong dependency on initial SOC stocks and the difficulty of reversing emission trends, highlighted the insufficiency of looking at carbon in isolation. This understanding led to the methodological change in **Chapter 5**, where coupling soil modelling with LCA highlighted that, in fact, "maximizing sequestration" can result in an increase in eutrophication, acidification and other environmental impacts. Among the findings, the inorganic fertilizers production emerged as one of the main sources (25-29%) of impact, opening the path for the exploration of alternative production configurations. This led to the exploration of other, closed-loop nutrient sources in **Chapter 6**, which shifted the boundary to sewage sludge recovery. Ultimately, realizing that closed-loop solutions can also harbour "hidden" energy implications, **Chapter 7** applied a prospective LCA to newly developing technologies in resource recovery (HPHTE), closing the loop by showing how "early stages" in eco-design can solve the trade-offs observed in previous chapters. The entire sequence therefore confirms that sustainability can neither be defined by one single indicator, such as carbon sequestration or circularity rate, but rather requires a multi-dimensional assessment tool.

8.1 SUMMARY OF MAIN FINDINGS

This doctoral research developed an integrated assessment framework for carbon mitigation in agriculture using the concepts of Soil Organic Carbon (SOC), Life Cycle Assessment (LCA), and Circular Economy (CE) models. Looking back on the development approach from systematic review, spatial modelling, environmental impact assessment, and emerging technology valorisation, it is evident that there are both potential and limitations in using Carbon Farming for achieving sustainability goals.

The systematic review established that carbon sequestration rates in staple crops range from very low (0-0.5 Mg C ha^{-1} yr^{-1}) to medium values (1-5 Mg C ha^{-1} yr^{-1}), with wheat-maize rotations achieving 4.96 Mg C ha^{-1} yr^{-1}. However, absence of standardized terminology for reporting CS rates significantly hampers cross-study comparison and undermines Carbon Farming credibility in policy and market contexts. The findings confirm that Carbon Farming has mitigation potential to be exploited and highlight the need for standardization in the sector both in terms of quantification and in terms of terminologies.

A regional modelling with RothC-26.3 in Flanders showed that agricultural soils in the region are net sources of carbon emissions, with peaks over 20 years of 9 t C ha^{-1}. Carbon Farming practices can only partly balance these emissions: cover cropping reduces emissions by around 20% (up to - 6.5 t C ha^{-1}), while rotation improvement has a mixed impact depending on baseline levels of carbon inputs. The positive correlation to the initial SOC stocks, shown in Pearson $r = -0.78$, highlights that initial high levels of SOC counterintuitively limit soil C sequestration. This finding has critical importance for the CRCF implementations is regions with similar pedoclimatic zones. Farmers managing high SOC soils may face limited economic revenues from carbon credit schemes unless payment structures account for SOC variability in baselines. These results informed the joint LCA-SOC study in Northen Italy where baseline conditions are more uniform than in Flanders and may be better analysed also in a emission reduction context.

The key insights from the joint SOC-LCA assessment conducted in Northern Italy are that Carbon Farming practices result in a notable increase in soil C stocks, but at the same time, these practices also produce environment-related trade-offs for different impact categories. Farmyard manure practices allow for the sequestration of 0.90 t CO_2 $ha^{-1}yr^{-1}$, but simultaneously increase acidification by 254%, marine eutrophication by 372%, and 243% for terrestrial eutrophication. In contrast, reduced tillage practices or those involving cover crops result in improved environmental performances with moderate C sequestered at 0.25-32 t CO_2 $ha^{-1}yr^{-1}$, at the same time lowering eutrophication impacts. Notably, soil carbon sequestration is a key indicator for climate mitigation but the study demonstrates the necessity of comprehensive systems-level sustainability assessment beyond carbon, accounting for multiple impact categories. The study highlighted inorganic fertilizers production as one of the main impacts' sources (25-29%), opening the path for the exploration of alternative production configurations. For this reason, the fourth study, regarding the use of circular inputs for fertilizers production, was carried out.

Circular economy assessments revealed simultaneous trade-offs between the social dimension and the environmental one. Phosphorus recovery from sewage sludge ash demonstrates modest environmental benefits in specific configurations (LG-MgO-based precipitants outperforming other circular alternatives). The Social Hotspots Analysis reveals medium-high social risks in upstream conventional mineral phosphorous extraction supply chains, particularly Egyptian phosphorite mining. From this study it emerges also the necessity to apply circular economy principles not only to the inputs of the agri-food sector but also to the outputs, with wasted deserving particular attention. This is why a prospective life cycle assessment of high-pressure high-temperature extraction (HPHTE) for olive pomace valorisation was carried out to validate technology scalability: HPHTE ensures 74% greenhouse gas reduction at pilot scale while reducing energy demand by 83% and water consumption by 93% when compared with conventional extraction. These findings underscore that single-metric optimization (carbon or resource recovery) inadvertently generates environmental and social externalities elsewhere in the product system.

Four methodological contributions advance Carbon Farming crediting frameworks and agricultural sustainability assessments: (1) coupled soil-LCA assessment should standardize within EU Carbon Removal Certification Framework to capture environmental trade-offs beyond carbon metrics; (2) spatial heterogeneity requires mandatory geographically explicit modelling and validation (RothC achieved $R^2 = 0.52$ in Flanders); (3) terminology standardization is essential-adoption of Mg C ha^{-1} yr^{-1} functional units and explicit carbon pool documentation should be mandatory in crediting schemes; (4) prospective LCA for emerging technologies enables early-stage hotspot identification and design optimization before industrial scaling.

The CRCF framework foresees the possible generation of carbon credits from Carbon Farming practices. Findings suggest that, given the magnitude, the temporal horizon and the risk of reversal of carbon sequestration the best option for the implementation of Carbon Farming practices should be for in-setting purposes rather than off-setting. Also, barriers observed in farmers willingness to adopt such practices is low, due to the fact that Carbon Farming requires initial investments which are seen as a “tax” without a proper explanation of the co-benefits, including long-term fertility with savings in the use of fertilizers and inputs which in turn can translate into economic benefits.

Based on the evidence gathered, particularly the reversibility risks identified in Flanders and the nutrient trade-offs observed in Northern Italy, this thesis suggests a strategic pivot for the implementation of the EU Carbon Removal Certification Framework. Given the high variability of soil dynamics, Carbon Farming credits are ill-suited for pure "off-setting" schemes (where emission rights are sold to external sectors), as this poses significant risks of greenwashing and non-permanence. Instead, policy mechanisms should prioritize "in-setting" approaches, where carbon removals are used to reduce the Scope 3 emissions within the agri-food supply chain itself. This approach ensures that the physical reality of the carbon cycle remains linked to the agricultural product, reducing the systemic risk associated with reversal. Additionally, the narrative around the concept of Carbon Farming needs to change from a "compliance cost" to a "business opportunity." As of today, environmental practices are seen as a "tax" on agricultural production. Yet, the agronomic data that has been reviewed in this work indicates that the development of SOC needs to be seen first and foremost as a "value" for the resilience of the farm system, rather than a "commodity" that is sold on the carbon markets. New policy instruments need to reward this co-benefit. By changing the narrative around the development of SOC from a cost of carbon markets to a cost of production capital, the engagement of the agricultural community with the concept of Carbon Farming can be delinked from the price volatility of carbon credits.

8.2 FUTURE PERSPECTIVES

Policy frameworks must evolve to reflect the environmental and social trade-offs identified in this research. Common Agricultural Policy payments and voluntary carbon market standards should incorporate multi-criteria assessment methodologies, explicitly weighting acidification, eutrophication, and water use alongside carbon sequestration. The demonstrated spatial and management-specific variability in carbon responsiveness indicates that standardized regional payment rates may inadvertently create perverse incentives, particularly in areas with high baseline productivity where carbon sequestration potential is inherently limited.

Social hotspot analysis findings highlight that European carbon removal strategies cannot ignore upstream supply chain impacts. As the EU scales circular economy transitions through mechanisms such as the Carbon Removal Certification Framework and extended producer responsibility schemes, the social performance of critical raw material suppliers becomes a direct measure of European sustainability commitments. Policy instruments should incorporate social due diligence requirements and preferentially support transition pathways that substitute critical raw materials with secondary sources (e.g., phosphorus recovery from sewage sludge ash) to reduce geopolitical risk and enhance social equity in extractive supply chains.

Significant limitations constrain current generalizability. RothC validation achieved $R^2 = 0.52$, reflecting the simplified representation of complex biogeochemical pathways. Climate stability assumptions underpin all scenarios, yet projected precipitation intensification and heat stress will fundamentally alter soil carbon dynamics over crediting periods. Agronomic trade-offs, including yield penalties of 2.6-7.6% for reduced tillage, remain incompletely characterized at regional scales. Circular economy scaling economics present unresolved constraints-technological feasibility and environmental preferability do not guarantee market viability without resolution of capital costs and feedstock availability.

Despite the limitations regarding data granularity and model uncertainty (e.g., RothC $R^2 = 0.52$), the directional findings of this research remain robust and actionable. Sensitivity analyses carried out on these case studies verify that although the absolute values of sequestration or impacts may change due to climatic changes, the relative position of the management practices remains the same. For example, the trade-off between organic fertilizers and nutrient leaching, as discussed in Chapter 4, holds true in different sensitivity analyses, thereby reinforcing the need for caution in the indiscriminate use of manure application. Again, though the HPHTE assessment in Chapter 6 is based on pilot-scale data, the magnitude of energy savings over conventional methods is so high that investment in scaling up this technology is justified. Therefore, the identified "hotspots" and "trade-offs" serve as reliable guardrails for policy design, even in the absence of perfect datasets. A methodological priority for future work is the full propagation of SOC modelling uncertainty, including the limited explanatory power of RothC in certain pedoclimatic contexts, into LCA results. This would enable a truly integrated uncertainty analysis that accounts for both the variability of carbon sequestration estimates and the sensitivity of life cycle impacts to those estimates.

Future research should explicitly incorporate biochar and compost as carbon farming strategies within the integrated SOC modelling and LCA framework developed in this thesis. Biochar, in particular, represents a high-priority area given its recognised potential as a negative emission technology and its growing relevance in EU climate policy, including the Carbon Removal Certification Framework (CRCF). The integration of biochar into the RothC model through dedicated parameterisation schemes (e.g., the addition of a stable, low-decomposition pool) would allow direct comparison with the cover crop, farmyard manure, and reduced tillage scenarios assessed in Chapter 5, enabling a comprehensive trade-off analysis that includes both long-term sequestration permanence and life-cycle environmental impacts (e.g., upstream emissions from feedstock collection, pyrolysis energy demand, and transport). Similarly, the substitution of mineral fertilisers with mature compost - as a complement or alternative to the farmyard manure scenario - could be assessed in terms of both SOC dynamics and multi-impact LCA performance, particularly regarding eutrophication and acidification trade-offs. Such analyses would directly address the current gap in combined biochar/LCA studies and contribute to the methodological standardisation required for crediting and MRV purposes.

Furthermore, future work should extend the methodological framework developed here towards a full sustainability assessment by integrating Life Cycle Costing (LCC) with the existing E-LCA and S-LCA analyses. Such integration would enable the evaluation of the economic feasibility of carbon farming practices and circular fertiliser strategies, including cost-benefit analyses under different carbon credit price scenarios and accounting for MRV transaction costs, thereby completing the environmental-social-economic sustainability tripod.

Future research should prioritize: (1) longitudinal field validation campaigns spanning 5-10 years across contrasting soil and climatic zones to enhance model parameterization and validate spatial scaling; (2) development of integrated agronomic-LCA decision support tools for agricultural extension services; (3) supply chain mapping and social risk assessment protocols for critical raw material sourcing; (4) prospective technology assessment frameworks for agrifood waste valorisation; (5) regulatory framework harmonization across EU member states to enable broader farmer participation in carbon crediting schemes; (6) full sustainability assessment.

Carbon sequestration in soils represents a necessary but insufficient alone mitigation strategy for agricultural climate commitments. Achieving European sustainability goals requires systems-level assessment frameworks that explicitly integrate soil carbon dynamics, multi-impact life cycle assessment, and social sustainability across value chains. Future policy mechanisms must move beyond isolated carbon metrics toward holistic sustainability assessment as the scientific foundation for agricultural transition.

APPENDICES A and B

Appendices A and B

This chapter is extensively based on the following publication:

Arellano Vazquez, D. A., Gagliano, E., Del Borghi, A., Tacchino, V., **Spotorno, S**., & Gallo, M. (2024). *Carbon farming of main staple crops: A systematic review of carbon sequestration potential;* ***https://doi.org/10.3390/su16187907***

Appendix A

This appendix is referred to Chapter 3.

Table A- 1: CS rates of maize, wheat, and rice, reported in experimental studies based on analytical methods.

Main Crop of Study	Methodology Used to Declare the Carbon Stock	Time of Experiment (years)	Method Used to Calculate the Carbon Sequestration	CS Rates, Min and Max Values	Ref.
Maize (Zea mays L.)	Soil samples were taken in April 2012 (before sowing) and 2015 (after harvest) and processed using an elemental analyzer. Soil sample depth: 0.2 m. SOC initial = 9.0 g kg^{-1}. SOC stock initial = in Mg ha^{-1}	4	SOCSR = (Cstock1 - Cstock2)/Obs. period. SOCSR units = Mg C ha^{-1} yr^{-1}	min: -0.61 ± 0.38; max: 1.02 ± 0.44	(Dong et al., 2022)
Maize (Zea mays L.) and wheat (Triticum aestivum)	Soil samples were taken before sowing in 2003 and July 2010, then processed using an elemental analyzer and isotope ratio spectrometer. Soil sample depth: 0.2 m in 2003 and 1 m in 2010. SOC initial = 11.4 g kg^{-1}. SOC stock initial = in Mg ha^{-1}	7	SOCSR = (Cstock1 - Cstock2)/Obs. period. SOCSR units = Mg C ha^{-1} yr^{-1}	min: not declared; max: 0.184 ± 86	(Cong et al., 2015)
Rice (Oryza	Soil samples were taken	33	CSR = (Cstock1 -	min: 0.12; max: 0.2	(Hu et

sativa L.), wheat (Triticum aestivum L.)	every year after crops were harvested and before soil plowing, then processed using potassium dichromate and an external heating method. Soil sample depth: 0.2 m. SOC initial = 15.9 g kg^{-1}. SOC stock initial = 38 Mg ha^{-1}		Cstock2)/Obs. period. CSR units = Mg C ha^{-1} yr^{-1}		al., 2018)
Wheat (Triticum aestivum L.)	Soil samples were taken in June 2009 and processed by Walkley-Black method. Soil sample depth: 0-0.5, 0.5-0.10, 0.10-0.20, 0.20-0.30, 0.30-0.40, 40-50, and 50-60 m. SOC initial = in g kg^{-1}. SOC stock initial: CT = 45.1 Mg ha^{-1}, NT = 45.4 Mg ha^{-1}	17	Annual SOC sequestered = (Cstock1 - Cstock2)/Obs. period. Annual SOC sequestered units: Mg C ha^{-1} yr^{-1}	min: 0.05; max: 0.3	(Liu et al., 2014b)
Maize (Zea mays L.) and wheat (Triticum aestivum)	Soil samples were taken in June 2021 and processed by Walkley-Black method. Soil sample depth: 0.2 m. SOC initial = 9.52 g kg^{-1}. SOC stock initial = in Mg ha^{-1}	6	SOC sequestration = (aboveground straw residues + belowground straw residues). SOC sequestration units: Mg C ha^{-1} yr^{-1}	min: 2.583; max: 3.801	(Liu et al., 2022)

Rice and wheat	Soil samples were taken every year after rice harvest and processed using vitriol acid-potassium dichromate oxidation. Soil sample depth: 0.2 m. SOC initial = 9.05 g kg^{-1}. SOC stock initial = in Mg ha^{-1}	10	-	min: -0.25; max: 0.52	(Fan et al., 2015)
Rice and wheat	Soil samples were taken every year after the rice harvest and processed afterward by vitriol acid-potassium dichromate oxidation. Soil sample depth: 0.2 m. SOC initial = 9.52 g kg^{-1}. SOC stock initial = in Mg ha^{-1}	34	Annual SOC sequestered = Cstock1 - Cstock2/Obs. period. Unit of measure: Mg C ha^{-1} yr^{-1}	min: 0.04; max: 0.64	-
Rice (Oryza sativa L.) and wheat (Triticum aestivum L.)	Soil samples were taken after the harvesting of wheat in 2008 and processed by Walkley-Black method. Soil sample depth: 0-0.15, 0.15-0.30, 0.30-0.45, and 0.45-0.60 m. SOC initial = 2.1 g kg^{-1}. SOC stock	9	Annual SOC sequestered = Cstock1-Cstock2/Obs. period. Unit of measure: Mg C ha^{-1} yr^{-1}	min: 0.33; max: 0.69	(Brar et al., 2013)

	initial = 5.9 Mg ha^{-1}				
Wheat and maize	Soil samples were taken after the harvesting of wheat in 1997 and 2009 and processed by potassium dichromate and external heating method followed by titration with ferrous ammonium sulfate. Soil sample depth: 0-0.2, 0.2-0.4, 0.4-0.6, and 0.66-1 m. SOC initial = for min values, 13.94 g kg^{-1}; for max, 15.6. SOC stock initial = for min values, 39.3; for max values, 44.5 Mg ha^{-1}	25	Annual SOC sequestered = Cstock1 - Cstock2/Obs. period. Unit of measure: Mg C ha^{-1} yr^{-1}	min: -7.18; max: 4.96	(Seremesic et al., 2017)

Table A- 2: CS reported conditions in experimental studies based on analytical methods.

Main Crop of Study	**Cropping System and Irrigation**	**Carbon Farming Practices to Evaluate (Tillage Reduction, Cover Cropping, Alternative Fertilizing)**	**Soil Profile at the Beginning and Pedoclimatic Conditions**	**Carbon Losses, Carbon Emissions, or Other Emissions Considered**	**Ref.**

Maize (Zea mays L.)	Crop rotation: No; Tillage: Conventional; Irrigation: No	Reduce the inorganic fertilizer: Evaluate the combination of animal manure (AM) with inorganic fertilizer.	Type of soil: Luvisol (FAO classification); Mean annual temperature (MAT): 7.5 °C; Annual precipitation (AP): 680 mm	N_2O emissions	(Dong et al., 2022)
Maize (Zea mays L.) and wheat (Triticum aestivum)	Crop rotation: Wheat-maize, wheat-fava bean (Vicia faba), maize-fava bean; Intercropping: Maize-wheat, wheat-fava bean, maize-fava bean; Tillage: Conventional; Irrigation: Yes	Enhancing the crop system: A comparison between intercropping and crop rotation.	Type of soil: Sandy loam; MAT: 8.9 °C; AP: 168 ± 8 mm	Not reported or considered	(Cong et al., 2015)
Rice (Oryza sativa L.), wheat (Triticu	Crop rotation: Rice-wheat; Tillage: Conventio	Reduce the inorganic fertilizer: Comparison of long-term organic	Type of soil: Albic Luvisol; MAT: 13 °C; AP: 1300 mm	Estimated carbon loss of 0.46 Mg C ha^{-1} yr^{-1}	(Hu et al., 2018)

m aestivum L.)	nal; Irrigation: No	manure or manure combined with inorganic fertilizers versus long-term application of inorganic fertilizer.			
Wheat (Triticum aestivum L.)	Crop rotation: No; Tillage: Conventional and no tillage; Irrigation: No	Reduce tillage: Conventional tillage (CT) versus no tillage (NT) with crop residue incorporation.	Type of soil: Silt loam under the USDA texture class; MAT: 10.7 °C; AP: 555 mm	Not reported	(Liu et al., 2014b)
Maize (Zea mays L.) and wheat (Triticum aestivum)	Crop rotation: Maize-wheat-soybean-wheat, soybean-wheat, and maize-wheat; Tillage: Reduce tillage and no tillage; Irrigation: Yes	Reduce tillage: Reduced tillage with crop residue incorporation (CT) versus no tillage (NT) with crop residue incorporation. Enhancing the crop system: Comparison of different crop rotations.	Type of soil: Silt loam under the USDA texture class; MAT: 13.1 °C; AP: 555 mm	Loss pool from mineralization (47.2-51.5%) in comparison with annual biomass input reported	(Liu et al., 2022)

Rice and wheat	Crop rotation: Rice-wheat-rice-rape; Tillage: Reduce tillage; Irrigation: Yes	Reduce the inorganic fertilizer: Comparison of long-term organic manure or manure combined with inorganic fertilizers versus long-term application of inorganic fertilizer.	Type of soil: Acid purple soil; MAT: 17.5 °C; AP: 1290 mm	SOC decomposition rates of 0.20 Mg C ha^{-1} yr^{-1}	(Fan et al., 2015)
Rice and wheat	Crop rotation: Rice-wheat; Tillage: Reduce tillage; Irrigation: Yes	Reduce the inorganic fertilizer: Comparison of long-term organic manure or manure combined with inorganic fertilizers versus long-term application of inorganic fertilizer.	Type of soil: Calcareous purple soil; MAT: 17.4 °C; AP: 930 mm	SOC accumulation rates of 0.0055 Mg C ha^{-1} yr^{-1}	-
Rice (Oryza sativa L.) and wheat (Triticum	Crop rotation: Rice-wheat; Tillage: Not specified;	Reduce the inorganic fertilizer: Comparison of twelve combinations of organic and	Type of soil: Loamy sand soil; MAT: Not specified; AP: Not specified	Losses not reported	(Brar et al., 2013)

aestivum L.)	Irrigation: Yes	inorganic fertilizers, as well as residue integration.			
Wheat and maize	Crop rotation: For the min values, maize-wheat; for the max values, sugar beet-winter wheat-maize-spring barley; Tillage: For the min values, conventional tillage; for the max values, no tillage; Irrigation: Yes	Enhancing the crop system: Assess the changes in SOC stock in relation to the carbon input from nine wheat-based cropping systems and untilled grassland.	Type of soil: Clay loam-textured; MAT: 12.3 °C; AP: 625 mm	SOC content depletion rate of 0.245 Mg C ha^{-1} yr^{-1}. This was compared with annual biomass input reported	(Seremesic et al., 2017)

Table A- 3: CS rates of maize, wheat, and rice, reported in experimental studies based on modeling methods.

Main Crop of Study	Model Used, Type of Data Used for Modeling and its Source	Time of Experiment (Years)	Method Used to Calculate the Carbon Sequestration	CS Rates, Min and Max Values
Maize (Zea mays L.)	Model: ARMOSA; Data required: SOC, daily maximum and minimum temperature, precipitation, and global solar radiation; Source of data: Previous collected experiments	21	SOCSR = (Cstock1 - Cstock2)/Obs. period. SOCSR units = Mg C ha^{-1} yr^{-1}	min: -0.317; max: 0.027
Maize (Zea mays L.)	Model: Not named; Data required: SOC, bulk density, and yield; Source of data: NASS, STATSGO2 database, long-term studies conducted in Iowa, and other	6	δSOCSR/δt = (SOC × Ksoc) + (NHC × Knhc). δSOCSR units = Mg C ha^{-1} yr^{-1}	min: 0.069; max: 0.454

	scientific studies			
Wheat and Rice	Model: MetaWin 2.1 software; Data required: SOC, bulk density, and management data; Source of data: twenty-six scientific articles listed in the document. Non-linear equations for bulk density	More than 3	ΔDSOC = (Dsoct - Dsoc0) - (D'soct - D'soc0)/t. δSOCSR units = Mg ha^{-1} yr^{-1}	min: 0.003; max: 0.53
Rice and Wheat	Model: Monte Carlo approach; Data required: SOC, and management data; Source of data: World Soils Reference database scientific experiment	20	ΔCi = ((Cit - Ci(t - 20))/20) × LAi. ΔCi units = Mg ha^{-1} yr^{-1}	min: 0.182; max: 0.433

Table A- 4: CS reported conditions in experimental studies based on modeling methods.

Main Crop of Study	**Cropping System and Irrigation**	**Carbon Farming Practices to Evaluate (Tillage Reduction, Cover Cropping, Alternative Fertilizing)**	**Soil Profile at the Beginning and Pedoclimatic Conditions**	**Carbon Losses, Carbon Emissions, or Other Emissions Considered**
Maize (Zea mays L.)	Crop rotation: Maize monocropping and maize-winter wheat-soybean; Tillage: No tillage, ploughing, and vertical tillage; Irrigation: Yes	Reduce tillage: Conventional tillage (CT) versus no tillage (NT). Enhancing the crop system: Comparison of different crop rotations	Type of soil: Sandy loam; Mean annual temperature (MAT): 12.9 °C; Annual precipitation (AP): From 185.2 mm	Soil mineralization coefficient
Maize (Zea mays L.)	Crop rotation: Maize monocropping; Tillage: No tillage, and conventional tillage; Irrigation: Yes	Reduce tillage: Conventional tillage (CT) versus no tillage (NT)	Type of soil: Brandt silty clay loam; Mean annual temperature (MAT): Not declared; Annual precipitation (AP): From 772 mm	Soil mineralization
Wheat and Rice	Crop rotation: Wheat-rice-rice or wheat-rice; Tillage: No tillage, and conventional	Residue incorporation: Crop residue recycling. Reduce tillage:	Type of soil: Paddy soil, fluvo-aquic soil, and coastal solonchaks;	Not considered

	tillage; Irrigation: No	Conventional tillage (CT) versus no tillage (NT). Reduce the inorganic fertilizer: Animal manure applications	Mean annual temperature (MAT): 15 °C; Annual precipitation (AP): From 1250 mm	
Rice and Wheat	Crop rotation: Rice-Wheat; Tillage: No tillage, and conventional tillage; Irrigation: Yes	Reduce tillage: Conventional tillage (CT) versus no tillage (NT). Reduce the inorganic fertilizer: Input reduction	Type of soil: Sandy loams to silty clay loam; Mean annual temperature (MAT): Variable; Annual precipitation (AP): Variable	Global warming potential

Appendix B

Appendix B, **Figure B1**, displays the trend of crop-related publications by year, with a steady increase from thirteen publications in 2001 to 200 publications in 2022. There was a notable increase from 86 to 145 documents published (more than 40% more) between 2014 and 2015. Then, every year between 2015 and 2018 showed significant increases, with 121 documents in 2016, 141 documents in 2017, and 211 documents in 2018. Notable events may have influenced the incremental tendency such as the Paris Agreement signatures and the formal adoption of the SDGs. Notably, since the Paris Agreement was signed and the SDGs were adopted in 2015, the number of publications for sequestration and storage has increased by 230%, maize by 260%, wheat by 156%, and rice by 219%.

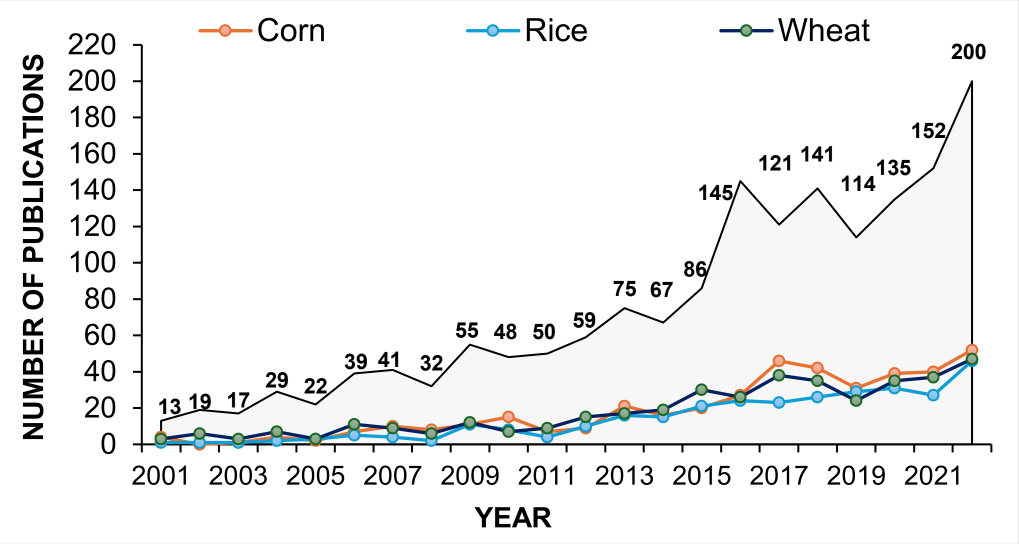


Figure B- 1: Number of publications about carbon sequestration in 2001-2022 period.

Appendix B, **Figure B2**, displays the top ten subject areas, where environmental science and agricultural/biological areas account for ~66% of all documents published in Carbon Farming. Business management and accounting also rank among the top ten areas of interest in the database of individual crops of interest, because CS by main staple crops is viewed not only as an environmental solution, but also as a business opportunity in the broader economics, econometrics, and finance areas (Xia et al., 2018).

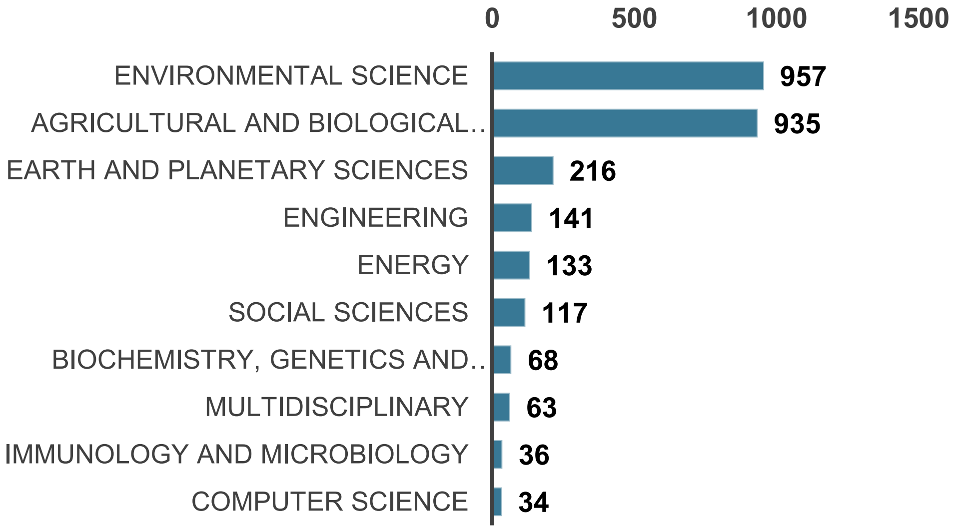


Figure B- 2: Top ten subject interest areas for carbon sequestration publications (2001-2022).

The USA, Brazil, Canada, and India are among the top ten countries in scientific production, accounting for 33.1% of the total documents published, Appendix B, **Figure B2**, but also being among the top three main staple crop exporters in Appendix B, Table B1. China and Japan are among the top three staple crop importers but are also included in the list of the top ten in scientific document production, Appendix B, **Figure B3**. China, as one of the main importers of rice, is the main contributor to documents published on Carbon Farming related to rice, accounting for 28.4%. Contrarily, Japan, as one of the primary importers of maize, contributes with only 1% of documents published on Carbon Farming.

Table B - 1: World main traders of cereal staple crops (FAO, 2021).

	Maize	**Wheat**	**Rice**
Main exporters	Brazil, United States of America, and Argentina	Russian Federation, United States of America, Canada	India, Thailand, and Vietnam
Main importers	Japan, Mexico, and Vietnam	Indonesia, Egypt, and Turkey	Philippines, China, and Benin

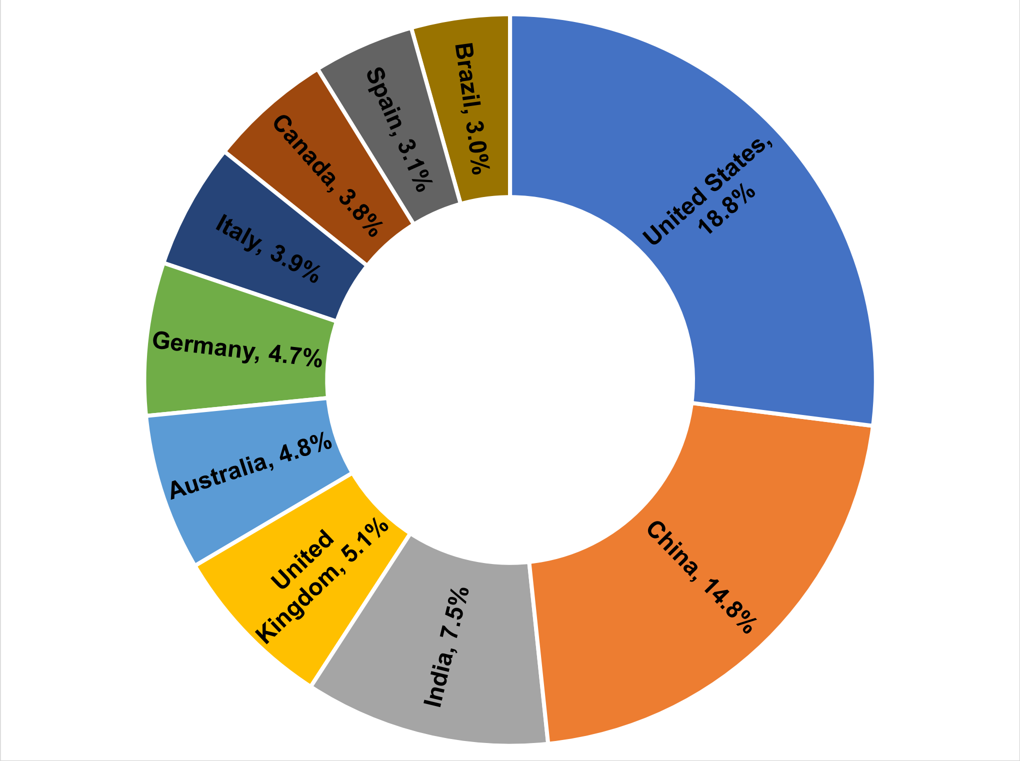


Figure B- 3: Top ten countries with carbon sequestration publications from 2001 to 2022.

APPENDIX C

Appendix C

This chapter is extensively based on the following publication, under review as of February 2026:

Esposito L., Gagliano E., Tacchino V., **Spotorno S**., Canziani R., El Chami D., Gallo M., Del Borghi A., Turolla A. (2026). *Environmental and social sustainability assessment of a circular process for the valorisation of sewage sludge ash and mining by-products into bio-based fertilisers* (*Journal of Cleaner Production*)

Appendix C

This appendix is referred to Chapter 6.

Table C - 1: Fraction of the assessed fertilisers (ENERGEO CV, ENERGEO CV TOP, LITHOZINC, PHEOSCOR) relative to the total granular fertiliser production in Italy in 2024 from TIMAC AGRO Italia S.p.A..

Fertilizer	Fraction
[-]	[%]
ENERGEO CV	1.1
ENERGEO CV TOP	1.5
LITHOZINC	1.5
PHEOSCOR	5.2

Description of Circular Economy (CE) scenario

Sewage sludge ash (SSA) is expected to be produced by four incineration plants located in Lombardy region (**Table C-2**), overall treating 330,000 tonnes/year of unsuitable sewage sludge (SS) for direct land spreading.

Table C - 2: Annual SS input, SSA output and average distance from the fertiliser production plant (Ripalta Arpina, Cremona, Italy) for each assumed incinerator in CE scenario.

Location	Input SS	Output SSA	Average distance
[-]	[tonnes/y]	[tonnes/y]	[km]
Parona (PV, Italy)	140,000	10,500	121
Corte Olona (PV, Italy)	55,000	4,100	42
Sesto San Giovanni (MI, Italy)	65,000	4,900	67
Como (CO, Italy)	70,000	5,300	116

After incineration, SSA will be moved to two phosphorus (P) storage and extraction sites, each one treating up to 15,000 tonnes/year of SSA. The northern one, potentially located at an industrial wasteland in Meda (Monza-Brianza, Italy), will manage SSA from northern incinerators (Sesto San Giovanni, Como), while the southern one, potentially located at an industrial wasteland in Gropello Cairoli (Pavia, Italy), will manage SSA from southern incinerators (Parona, Corte Olona).

P will be recovered through a wet acid extraction, using either HCl or H_2SO_4, and an alkaline precipitation, using commercial-grade $Ca(OH)_2$ or a low-grade magnesium oxide mining by-product (LG-MgO) supplied from the Zubiri quarry (Navarre, Spain) of the private mining company Magnesitas Navarras. Since LG-MgO was considered as a production residue, only the impacts related to its transport were accounted, in accordance with the Polluter Pays principle. **Table C-3** displays the ratio between the P-based output and SSA input for the four recovery configurations evaluated in CE scenario, based on pilot-scale data reported in Boniardi et al. (2024). In this study, the optimal P recovery configuration detailed in Esposito et al. (2024) was considered (**Table C-4**; HCl extraction - 0.8 neq/L, 10 L $_{\text{acid solution}}$/kg $_{\text{SSA}}$, 30 min; $Ca(OH)_2$ precipitation - 30% $_{\text{w/v}}$).

Table C - 3: Mass ratio between output P-based product and input SSA for the four assumed process configurations in CE scenario.

Extraction	**Precipitation**	**P-based product/SSA**
[-]	[-]	[%]
H_2SO_4	$Ca(OH)_2$	46
H_2SO_4	LG-MgO	63
HCl	$Ca(OH)_2$	46
HCl	LG-MgO	85

Table C - 4: Mass, volume and energy inputs and outputs from the optimal P recovery process configuration described in Esposito et al. (2024) P (Prec).

Flows	**U.M.**	**Value**
Input		
SSA	[kg/kg P_{rec}]	14
Acid extractant	[L/kg P_{rec}]	10
Water	[L/kg P_{rec}]	152
Anionic polyelectrolyte	[g/kg Prec]	2
Alkaline precipitant	[kg/kg P_{rec}]	5
Electric energy	[kWh/kg P_{rec}]	2
Output		

Acidic solid residues	[kg/kg P_{rec}]	13
Supernatant	[L/kg P_{rec}]	158
Precipitate	[kg/kg P_{rec}]	8

Table C - 5: Fraction of raw materials needed to produce the assessed fertilisers (ENERGEO CV, ENERGEO CV TOP, LITHOZINC, PHEOSCOR) in BAU and CE scenarios. Fractions are referred to 1 tonne of fertiliser, packaging included. P-based recovered products are named as "acid extractant - alkaline precipitant".

Fertiliser - scenario	**Component**	**Fraction**
[-]	[-]	[%]
ENERGEO CV - BAU	Defecation lime from beet sludge	95
	Gypsum	30
	Pheoflore Rinfusa	1
	Oil	0.3
ENERGEO CV - CE	HCl - $Ca(OH)_2$	24
	CoProB - Defecation lime from beet	60
	White gypsum	30
ENERGEO CV TOP - BAU	Defecation lime from beet sludge	60
	Gypsum	40
	Pheoflore Rinfusa	1

	Oil	0.3
	Phosphorite	11
ENERGEO CV TOP - CE	H_2SO_4 - $Ca(OH)_2$ Phosphorite	17 7
	CoProB - Defecation lime from beet	62
	White gypsum	30
LITHOZINC - BAU	D-Coder IBC 1 T Ammonium sulphate 21%	0.5 15
	Zinc sulphate mono Big-Bag 1250 kg	0.3
	Talc	0.4
	Urea 46	4.5
	Potassium chloride standard	23.5
	Diammonium phosphate (DAP) 18-46-0	5
	Oil	0.4
	Complex single superphosphate (SSP) Rinfusa RIP	56
LITHOZINC - CE	Zinc sulphate Potassium chloride	0.3 26.5
	Ammonium sulphate 21%	9
	TOP PHOS	42
	HCl - LG-MgO	10
	Urea	4
	Diammonium phosphate (DAP)	13
PHEOSCOR - BAU	Magnesium oxide Sack 1 T Pheoflore IBC 1 T	2.3 1
	Heptahydrate iron sulphate Big-Bag 1500 kg	2
	Zinc sulphate mono Big-Bag 1250 kg	0.027
	Talc	0.4
	Pentahydrate borax Sack 1 T	0.1
	Dolomite	1

	Oil	0.4
	Complex single superphosphate (SSP) Rinfusa RIP	99.5
PHEOSCOR - CE	Iron sulphate	1.6
	H_2SO_4 - LG-MgO	7
	TOP PHOS	95
	Magnesium oxide	2.2
	Etibor - Ethanolamine	0.05

Table C - 6: Primary and secondary packaging mass and average distance between packaging provider sites and fertiliser manufacturing plant. Packaging materials are indicated between parentheses.

Packaging (material)	**Mass**	**Distance**
[-]	[kg]	[km]
Big Bag (PP)	36,113	2,113
Big Bag (PP)	16,477	666
Big Bag (PP)	95,500	566
PPH plastic bags (LDPE)	97,839	55
PPH plastic bags (LDPE)	71,417	55
Plastic film for silos storage (LDPE)	38,216	55
Coverstretch (PVC)	28,145	1,362
Pallet (wood)	249,546	150

Table C - 7: Annual natural gas consumption allocated to the boiler (B) and cogeneration (C) units.

Unit	**Natural gas consumption**
[-]	[Std m^3]
B1	2,050,185
B2	57,969
B3	30,066
B4	23,652

C1	1,664,113

Table C - 8: Electric energy purchased from and sold to the grid, and gross/net output to the cogeneration unit.

Data	Electric energy
[-]	[kWh]
Purchase	1,884,519
Sale	165,852
Gross production	6,110,328
Net production	6,022,893

Table C - 9: Emission allocation factors for the cogeneration unit. Fuel consumption is normalised to the input of primary energy to the cogeneration unit.

Parameter	Value
Electric efficiency	0.42
Thermal efficiency	0.9
Consumed fuel for electric energy production	0.88
Consumed fuel for thermal energy production	0.47
Consumed fuel for total energy production	1.35

Table C - 10: Gaseous emissions generated by the fertiliser manufacturing.

Gaseous compound	Emission
[-]	[tonnes/y]

SO_2	2.092
NO_x	37.46
Powders	3.824
VOC	12.23
CO	0.042

Table C - 11: Plant waste masses, CER codes, management routes and distances from management plant locations. Asterisks indicate wastes to be considered as hazardous.

CER code	**Management option**	**Management plant location**	**Mass**	**Distance**
[-]	[-]	[-]	[kg]	[km]
070213	Disposal	Fombio (IT)	11,200	19
080318	Disposal	Codogno (IT)	73	16
120112*	Disposal	Leno (IT)	640	54
130205*	Disposal	Leno (IT)	2,200	54
130802*	Disposal	Leno (IT)	1,560	54
150102	Recycling	Fombio (IT)	85,120	19
150102	Recycling	Lodi (IT)	27,330	24
150103	Recycling	Castelleone (IT)	139,320	10
150106	Disposal	Albino (IT)	1,190	68
150106	Disposal	Castelleone (IT)	63,620	10
150110*	Disposal	Leno (IT)	270	54
150202*	Disposal	Leno (IT)	600	54
150203	Disposal	Leno (IT)	120	54
150203	Disposal	Codogno (IT)	8,370	16
160107*	Disposal	Leno (IT)	5	54
160213*	Disposal	Codogno (IT)	10	16
160214	Disposal	Codogno (IT)	2,804	16
160305*	Disposal	Leno (IT)	300	54
160601*	Disposal	Leno (IT)	50	54

170107	Disposal	Gombito (IT)	690	1
170405	Recycling	Grumello Cremonese (IT)	55,580	17
170411	Recycling	Grumello Cremonese (IT)	460	17
170603*	Disposal	Gorlago (IT)	130	52
200121*	Disposal	Codogno (IT)	10	16
200304	Disposal	Liscate (IT)	16,680	43

Table C - 12: Results from the E-LCIA of ENERGEO CV production assessed in BAU and CE scenarios.

Impact sub-category	**U.M.**	**BAU**			**CE**		
		Upstream	**Core**	**Total**	**Upstream**	**Core**	**Total**
GWPfos	[kg CO_2 eq]	4.89E+01	7.08E+01	1.20E+02	3.55E+02	7.08E+01	4.26E+02
GWPbio	[kg CO_2 eq]	2.82E+00	7.30E-01	3.55E+00	1.06E+01	7.20E-01	1.13E+01
GWPlul	[kg CO_2 eq]	1.30E-01	0.00E+00	1.30E-01	2.00E-01	0.00E+00	2.00E-01
GWPtot	[kg CO_2 eq]	5.19E+01	7.15E+01	1.23E+02	3.66E+02	7.15E+01	4.37E+02
ODP	[kg CFC-11 eq]	1.45E-05	7.30E-07	1.52E-05	3.58E-05	7.30E-07	3.65E-05
AP	[mol H^+ eq]	4.10E-01	9.00E-02	5.00E-01	2.47E+00	9.00E-02	2.56E+00
EPfw	[kg P eq]	2.54E-03	1.34E-04	2.67E-03	3.59E-02	1.34E-04	3.60E-02
EPma	[kg N eq]	6.00E-02	1.50E-01	2.10E-01	1.05E+00	1.50E-01	1.20E+00
EPte	[mol N eq]	6.20E-01	7.80E-01	1.40E+00	5.30E+00	7.40E-01	6.04E+00
POCP	[kg NMVOC eq]	1.50E-01	3.90E-01	5.40E-01	8.20E-01	3.80E-01	1.20E+00
ADPm	[kg Sb eq]	4.80E-04	2.52E-07	4.80E-04	2.99E-03	2.52E-07	2.99E-03
ADPf	[MJ - net calorific value]	1.95E+03	7.77E+01	2.03E+03	5.03E+03	7.67E+01	5.11E+03
WDP	[m^3]	3.44E+01	1.69E+00	3.61E+01	4.76E+02	1.66E+00	4.77E+02
PERE	[MJ]	2.91E+01	1.52E+01	4.43E+01	1.19E+02	1.51E+01	1.35E+02
PERM	[MJ]	1.42E+02	3.01E+00	1.45E+02	2.13E+02	3.01E+00	2.16E+02

PERT	[MJ]	1.71E+02	1.82E+01	1.89E+02	3.33E+02	1.82E+01	3.51E+02
PENRE	[MJ]	1.58E+03	7.52E+01	1.65E+03	4.56E+03	7.57E+01	4.63E+03
PENRM	[MJ]	3.62E+02	6.70E-03	3.62E+02	5.39E+02	6.70E-03	5.39E+02
PENRT	[MJ]	1.94E+03	7.52E+01	2.01E+03	5.10E+03	7.57E+01	5.17E+03
SM	[kg]	9.50E+02	0.00E+00	9.50E+02	4.20E+02	0.00E+00	4.20E+02
RSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
NRSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
FW	[m^3]	8.50E-01	4.00E-02	8.90E-01	1.11E+01	4.00E-02	1.11E+01
LU	[-]	6.40E+02	1.58E+01	6.56E+02	5.58E+03	1.58E+01	5.60E+03
HWD	[kg]	6.00E-02	1.40E-01	2.00E-01	8.10E-01	1.40E-01	9.50E-01
NHWD	[kg]	1.72E+00	2.00E-01	1.92E+00	9.04E+00	2.00E-01	9.24E+00
RWD	[kg]	1.86E+03	6.82E+01	1.93E+03	4.81E+03	6.87E+01	4.88E+03
PM	[disease inc]	3.25E-06	1.68E-06	4.93E-06	1.71E-05	1.71E-06	1.88E-05
IR	[kBq U235 eq]	2.32E+00	1.20E-01	2.44E+00	1.05E+01	1.20E-01	1.07E+01
ET	[CTUh]	1.09E+03	1.77E+01	1.10E+03	6.06E+03	1.82E+01	6.08E+03
HTc	[CTUh]	1.23E-08	1.04E-08	2.27E-08	5.20E-08	1.04E-08	6.24E-08
HTnc	[CTUh]	7.56E-08	3.75E-09	7.93E-08	4.55E-07	3.74E-09	4.58E-07

Table C - 13: Results from the E-LCIA of ENERGEO CV TOP production assessed in BAU and CE scenarios.

Impact sub-category	**U.M.**	**BAU**			**CE**		
		Upstream	**Core**	**Total**	**Upstream**	**Core**	**Total**
GWPfos	[kg CO_2 eq]	9.18E+01	7.45E+01	1.66E+02	2.78E+02	7.08E+01	3.49E+02
GWPbio	[kg CO_2 eq]	4.48E+00	7.70E-01	5.25E+00	4.89E+00	7.10E-01	5.60E+00
GWPlul	[kg CO_2 eq]	7.70E-01	0.00E+00	7.70E-01	5.40E-01	0.00E+00	5.40E-01
GWPtot	[kg CO_2 eq]	9.70E+01	7.53E+01	1.72E+02	2.83E+02	7.15E+01	3.55E+02

ODP	[kg CFC-11 eq]	1.72E-05	7.05E-07	1.79E-05	3.92E-05	7.05E-07	3.99E-05
AP	[mol H$^+$ eq]	7.20E-01	9.00E-02	8.10E-01	4.91E+00	9.00E-02	5.00E+00
EPfw	[kg P eq]	1.00E-02	0.00E+00	1.00E-02	3.00E-02	0.00E+00	3.00E-02
EPma	[kg N eq]	1.30E-01	1.60E-01	2.90E-01	1.08E+00	1.50E-01	1.23E+00
EPte	[mol N eq]	1.27E+00	7.80E-01	2.05E+00	5.81E+00	7.40E-01	6.55E+00
POCP	[kg NMVOC eq]	3.00E-01	4.10E-01	7.10E-01	1.07E+00	3.80E-01	1.45E+00
ADPm	[kg Sb eq]	0.00E+00	0.00E+00	0.00E+00	3.26E-04	2.52E-07	3.26E-04
ADPf	[MJ - net calorific value]	2.46E+03	8.17E+01	2.54E+03	5.98E+03	7.73E+01	6.05E+03
WDP	[m^3]	1.03E+01	1.77E+00	1.20E+01	7.36E+02	1.61E+00	7.37E+02
PERE	[MJ]	3.85E+01	1.60E+01	5.45E+01	7.38E+01	1.45E+01	8.82E+01
PERM	[MJ]	1.82E+02	3.13E+00	1.85E+02	1.42E+02	2.87E+00	1.45E+02
PERT	[MJ]	2.21E+02	1.91E+01	2.40E+02	2.16E+02	1.73E+01	2.33E+02
PENRE	[MJ]	2.04E+03	7.90E+01	2.12E+03	4.51E+03	7.34E+01	4.58E+03
PENRM	[MJ]	3.95E+02	6.70E-03	3.95E+02	1.24E+03	6.70E-03	1.24E+03
PENRT	[MJ]	2.44E+03	7.91E+01	2.52E+03	5.75E+03	7.34E+01	5.82E+03
SM	[kg]	6.00E+02	0	6.00E+02	2.98E+02	0	2.98E+02
RSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
NRSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
FW	[m^3]	6.10E-01	4.00E-02	6.50E-01	1.63E+01	3.67E-02	1.63E+01
LU	[-]	9.88E+02	1.55E+01	1.00E+03	5.55E+03	1.58E+01	5.57E+03
HWD	[kg]	6.00E-02	1.40E-01	2.00E-01	8.20E-01	1.40E-01	9.60E-01
NHWD	[kg]	3.22E+00	2.10E-01	3.43E+00	3.54E+00	1.90E-01	3.73E+00
RWD	[kg]	2.36E+03	7.17E+01	2.43E+03	5.81E+03	6.72E+01	5.88E+03

PM	[disease inc]	0.00E+00	0.00E+00	0.00E+00	3.06E-05	1.71E-06	3.23E-05
IR	[kBq U235 eq]	6.55E+00	1.10E-01	6.66E+00	1.32E+01	1.50E-01	1.33E+01
ET	[CTUh]	1.48E+03	1.82E+01	1.50E+03	1.11E+03	1.68E+01	1.13E+03
HTc	[CTUh]	1.99E-08	9.88E-09	2.98E-08	4.20E-08	9.88E-09	5.19E-08
HTnc	[CTUh]	1.01E-07	3.52E-09	1.04E-07	3.70E-07	3.52E-09	3.73E-07

Table C - 14: Results from the E-LCIA of LITHOZINC production assessed in BAU and CE scenarios.

Impact sub-category	**U.M.**	**BAU**			**CE**		
		Upstream	**Core**	**Total**	**Upstream**	**Core**	**Total**
GWPfos	[kg CO_2 eq]	4.42E+02	7.71E+01	5.20E+02	5.00E+02	7.70E+01	5.77E+02
GWPbio	[kg CO_2 eq]	5.25E+00	8.20E-01	6.07E+00	4.58E+00	8.10E-01	5.39E+00
GWPlul	[kg CO_2 eq]	1.90E+00	0.00E+00	1.90E+00	4.00E-02	0.00E+00	4.00E-02
GWPtot	[kg CO_2 eq]	4.50E+02	7.79E+01	5.28E+02	5.05E+02	7.78E+01	5.83E+02
ODP	[kg CFC-11 eq]	2.95E-05	9.40E-07	3.04E-05	2.50E-05	9.42E-07	2.59E-05
AP	[mol H^+ eq]	2.26E+00	9.00E-02	2.35E+00	2.47E+00	9.00E-02	2.56E+00
EPfw	[kg P eq]	2.00E-02	0.00E+00	2.00E-02	2.00E-02	0.00E+00	2.00E-02
EPma	[kg N eq]	5.00E-01	1.50E-01	6.50E-01	6.30E-01	1.50E-01	7.80E-01
EPte	[mol N eq]	5.39E+00	7.60E-01	6.15E+00	6.42E+00	7.70E-01	7.19E+00
POCP	[kg NMVOC eq]	1.38E+00	3.90E-01	1.77E+00	1.65E+00	3.90E-01	2.04E+00
ADPm	[kg Sb eq]	1.16E-03	2.71E-07	1.16E-03	1.75E-03	2.69E-07	1.75E-03

ADPf	[MJ - net calorific value]	1.09E+04	1.01E+02	1.10E+04	1.19E+04	9.80E+01	1.20E+04
WDP	[m^3]	4.65E+01	2.23E+00	4.88E+01	9.00E+01	2.22E+00	9.23E+01
PERE	[MJ]	9.06E+01	2.01E+01	1.11E+02	8.69E+01	2.00E+01	1.07E+02
PERM	[MJ]	8.93E+01	3.99E+00	9.33E+01	9.42E+01	3.90E+00	9.81E+01
PERT	[MJ]	1.80E+02	2.41E+01	2.04E+02	1.81E+02	2.39E+01	2.05E+02
PENRE	[MJ]	1.04E+04	1.07E+02	1.05E+04	1.15E+04	9.85E+01	1.16E+04
PENRM	[MJ]	4.14E+02	7.72E-03	4.14E+02	2.94E+02	7.70E-03	2.94E+02
PENRT	[MJ]	1.08E+04	1.07E+02	1.09E+04	1.18E+04	9.85E+01	1.19E+04
SM	[kg]	0.00E+00	0.00E+00	0.00E+00	1.75E+02	0.00E+00	1.75E+02
RSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
NRSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
FW	[m^3]	1.30E-01	5.00E-02	1.80E-01	2.23E+00	5.00E-02	2.28E+00
LU	[-]	1.17E+03	2.13E+01	1.19E+03	5.50E+02	2.08E+01	5.70E+02
HWD	[kg]	6.00E-02	1.40E-01	2.00E-01	9.00E-02	1.40E-01	2.30E-01
NHWD	[kg]	3.91E+00	2.10E-01	4.12E+00	3.76E+00	2.10E-01	3.97E+00
RWD	[kg]	1.05E+04	9.73E+01	1.06E+04	1.14E+04	8.84E+01	1.15E+04
PM	[disease inc]	2.30E-05	1.73E-06	2.47E-05	2.86E-05	1.75E-06	3.04E-05
IR	[kBq U235 eq]	2.40E+01	1.60E-01	2.42E+01	1.32E+01	1.60E-01	1.34E+01
ET	[CTUh]	2.11E+03	2.21E+01	2.13E+03	2.98E+03	2.09E+01	3.00E+03
HTc	[CTUh]	2.92E-07	1.20E-08	3.04E-07	2.58E-07	1.21E-08	2.70E-07
HTnc	[CTUh]	3.82E-07	4.49E-09	3.87E-07	5.20E-07	4.33E-09	5.24E-07

Table C - 15: Results from the E-LCIA of PHEOSCOR production assessed in BAU and CE scenarios.

Impact sub-category	**U.M.**	**BAU**			**CE**		
		Upstream	**Core**	**Total**	**Upstream**	**Core**	**Total**

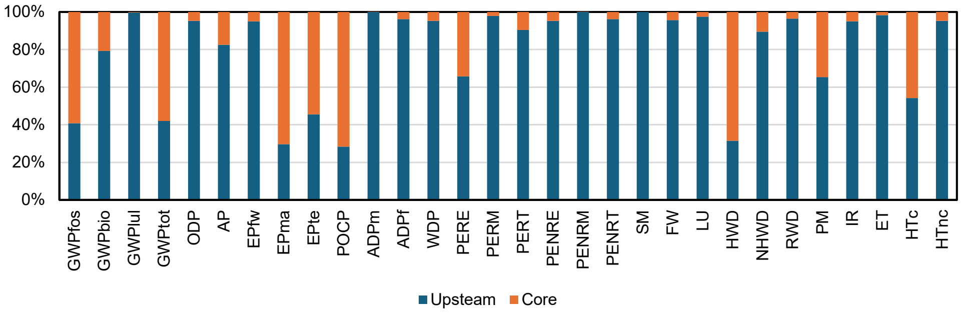
GWPfos	[kg CO_2 eq]	3.06E+02	8.17E+01	3.88E+02	3.02E+02	7.25E+01	3.75E+02
GWPbio	[kg CO_2 eq]	9.06E+00	8.66E-01	9.92E+00	2.33E+00	7.30E-01	3.06E+00
GWPlul	[kg CO_2 eq]	3.49E+00	0.00E+00	3.49E+00	2.00E-02	0.00E+00	2.00E-02
GWPtot	[kg CO_2 eq]	3.19E+02	8.25E+01	4.01E+02	3.05E+02	7.32E+01	3.78E+02
ODP	[kg CFC-11 eq]	3.40E-05	1.11E-06	3.51E-05	2.22E-05	7.34E-07	2.29E-05
AP	[mol H^+ eq]	2.20E+00	9.70E-02	2.30E+00	2.89E+00	9.00E-02	2.98E+00
EPfw	[kg P eq]	2.71E-02	2.06E-04	2.73E-02	4.15E-03	1.34E-04	4.28E-03
EPma	[kg N eq]	3.77E-01	1.50E-01	5.27E-01	5.00E-01	1.50E-01	6.50E-01
EPte	[mol N eq]	3.95E+00	7.62E-01	4.72E+00	5.16E+00	7.40E-01	5.90E+00
POCP	[kg NMVOC eq]	1.03E+00	3.90E-01	1.42E+00	1.39E+00	3.90E-01	1.78E+00
ADPm	[kg Sb eq]	1.72E-03	2.81E-07	1.72E-03	6.00E-04	2.53E-07	6.00E-04
ADPf	[MJ - net calorific value]	6.72E+03	1.09E+02	6.83E+03	7.80E+03	7.62E+01	7.88E+03
WDP	[m^3]	1.10E+02	2.62E+00	1.13E+02	1.90E+02	1.68E+00	1.91E+02
PERE	[MJ]	8.91E+01	2.36E+01	1.13E+02	4.66E+01	1.51E+01	6.17E+01
PERM	[MJ]	1.14E+02	5.12E+00	1.19E+02	6.91E+01	3.17E+00	7.23E+01
PERT	[MJ]	2.03E+02	2.88E+01	2.31E+02	1.16E+02	1.83E+01	1.34E+02
PENRE	[MJ]	6.44E+03	1.15E+02	6.56E+03	7.17E+03	7.49E+01	7.24E+03
PENRM	[MJ]	2.16E+02	8.46E-03	2.16E+02	5.79E+02	6.70E-03	5.79E+02
PENRT	[MJ]	6.66E+03	1.15E+02	6.77E+03	7.75E+03	7.49E+01	7.82E+03
SM	[kg]	0.00E+00	0.00E+00	0.00E+00	1.23E+02	0.00E+00	1.23E+02
RSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
NRSF	[MJ]	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00

FW	[m^3]	8.07E-01	5.90E-02	8.66E-01	4.50E+00	4.00E-02	4.54E+00
LU	[-]	1.84E+03	2.61E+01	1.86E+03	4.24E+02	1.60E+01	4.40E+02
HWD	[kg]	8.60E-02	1.36E-01	2.22E-01	1.10E-01	1.40E-01	2.50E-01
NHWD	[kg]	5.95E+00	2.16E-01	6.17E+00	6.10E-01	2.00E-01	8.10E-01
RWD	[kg]	6.47E+03	1.03E+02	6.57E+03	7.54E+03	6.80E+01	7.60E+03
PM	[disease inc]	2.14E-05	1.76E-06	2.31E-05	2.75E-05	1.73E-06	2.92E-05
IR	[kBq U235 eq]	2.97E+01	1.77E-01	2.99E+01	7.90E+00	1.20E-01	8.02E+00
ET	[CTUh]	2.97E+03	2.59E+01	2.99E+03	2.16E+03	1.82E+01	2.18E+03
HTc	[CTUh]	2.56E-07	1.32E-08	2.69E-07	2.66E-07	1.03E-08	2.76E-07
HTnc	[CTUh]	2.25E-07	4.79E-09	2.30E-07	3.48E-07	3.66E-09	3.52E-07

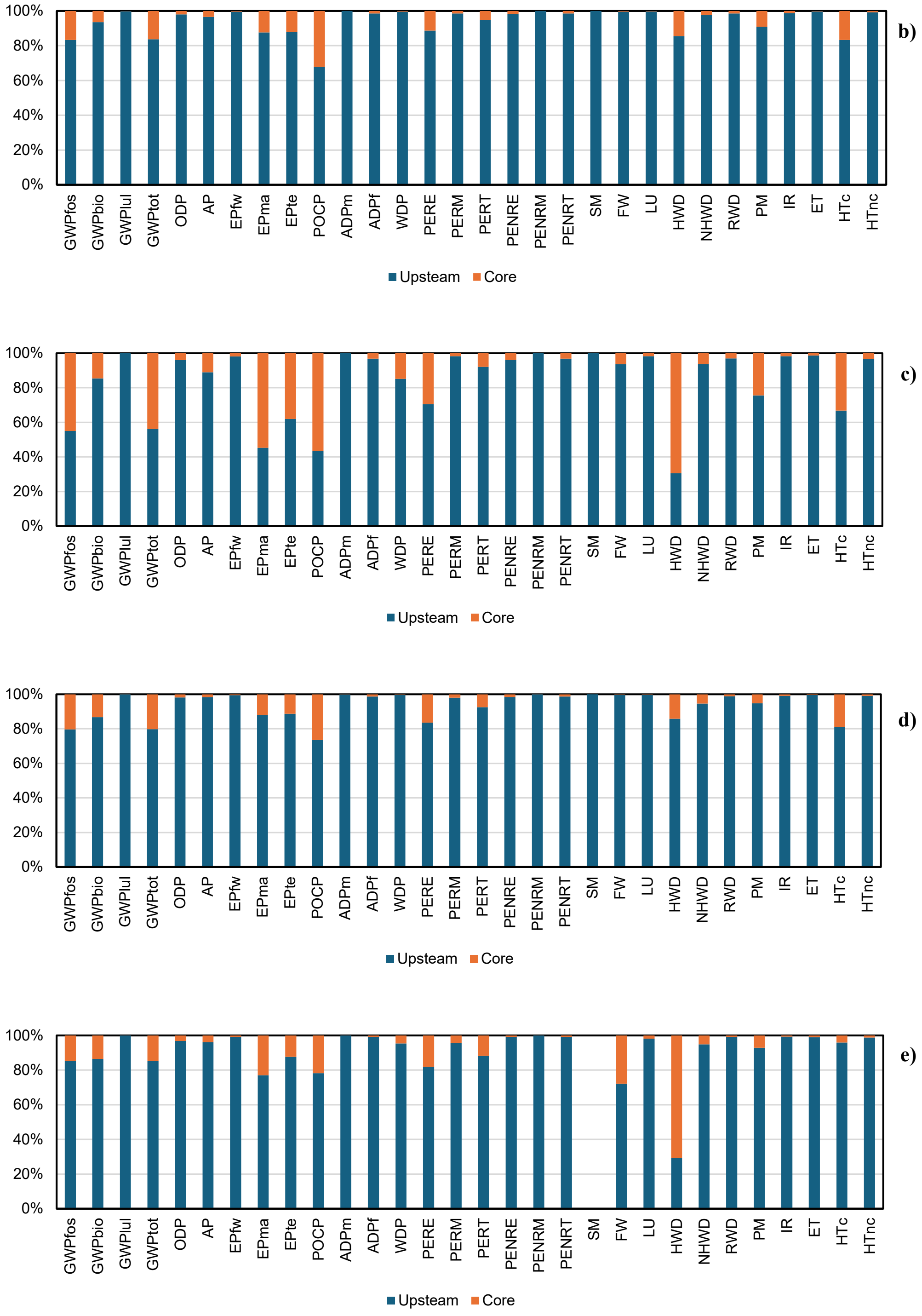
Table C - 16: Total environmental impacts of the four assessed configurations of P recovery from SSA. Recovery configurations are named as "extracting agent - precipitating agent".

Impact sub-category	**U.M.**	**H_2SO_4 - $Ca(OH)_2$**	**H_2SO_4 - LG-MgO**	**HCl - $Ca(OH)_2$**	**HCl - LG-MgO**
GWPfos	[kg CO_2 eq]	8.62E+02	4.09E+02	1.04E+03	6.00E+02
GWPbio	[kg CO_2 eq]	1.02E+01	9.34E+00	3.47E+01	3.39E+01
GWPlul	[kg CO_2 eq]	1.04E-01	9.54E-02	3.08E-01	3.00E-01
GWPtot	[kg CO_2 eq]	8.72E+02	4.20E+02	1.07E+03	6.19E+02
ODP	[kg CFC-11 eq]	1.64E-04	1.54E-04	1.04E-04	9.49E-05
AP	[mol H^+ eq]	2.23E+01	2.21E+01	6.52E+00	6.34E+00
EPfw	[kg P eq]	4.86E-02	4.83E-02	8.10E-02	8.07E-02
EPma	[kg N eq]	1.42E+00	1.43E+00	1.12E+00	1.14E+00

EPte	[mol N eq]	1.13E+01	1.15E+01	7.92E+00	8.03E+00
POCP	[kg NMVOC eq]	4.02E+00	3.94E+00	2.19E+00	2.12E+00
ADPm	[kg Sb eq]	2.25E-05	2.09E-05	1.20E-02	1.20E-02
ADPf	[MJ - net calorific value]	1.93E+04	1.86E+04	1.10E+04	1.03E+04
WDP	[m^3]	2.55E+03	2.56E+03	7.07E+02	7.12E+02



a)


b)
c)
d)
e)
100%
80%
60%
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20%
0%
GWPfos
GWPbio
GWPlul
GWPtot
ODP
AP
EPfw
EPma
EPte
POCP
ADPm
ADPf
WDP
PERE
PERM
PERT
PENRE
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Upsteam
Core

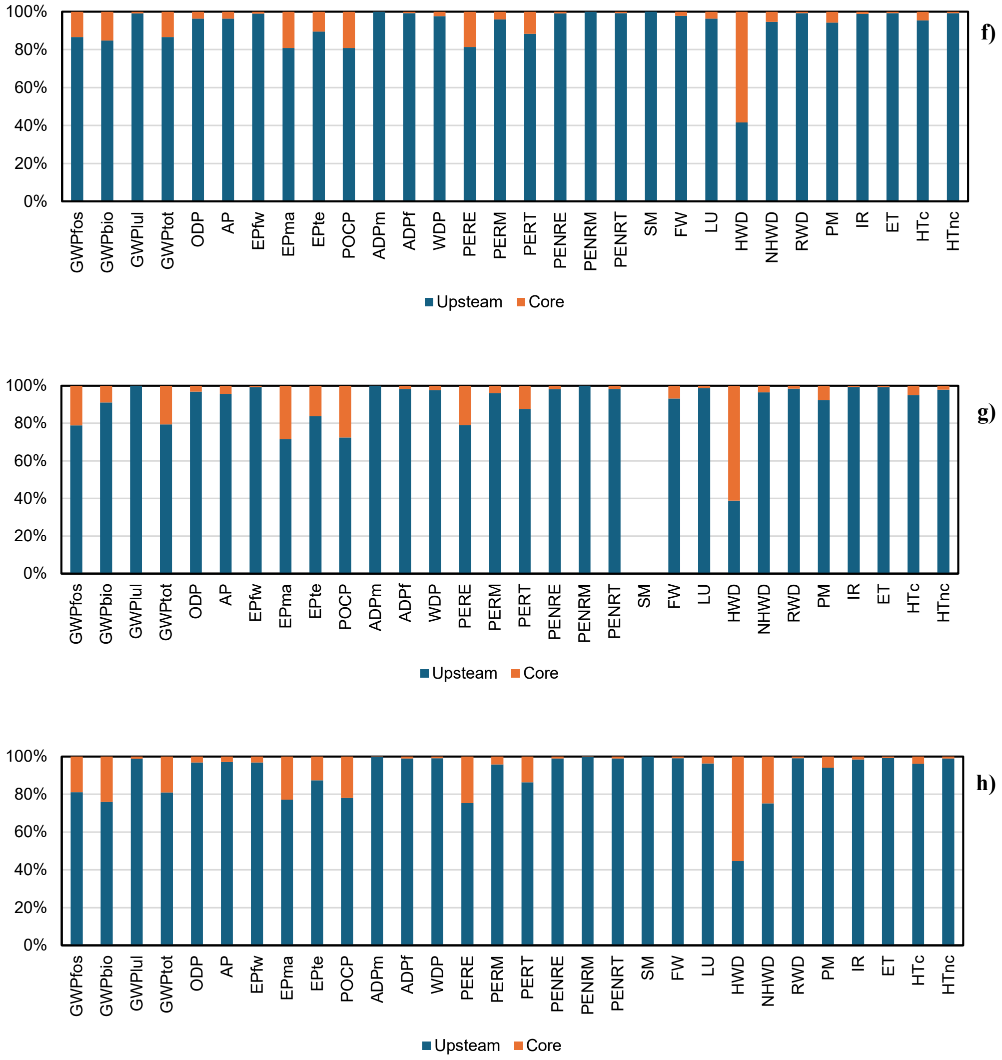

Figure C - 1: Upstream and core contributions in the assessed environmental impact categories for ENERGEO CV (a-b), ENERGEO CV TOP (c-d), LITHOZINC (e-f) and PHEOSCOR (g-h) production assessed in BAU (a, c, e, g) and CE (b, d, f, h) scenarios. Upstream and core contributions are individually normalised to the total impact for each sub-category.

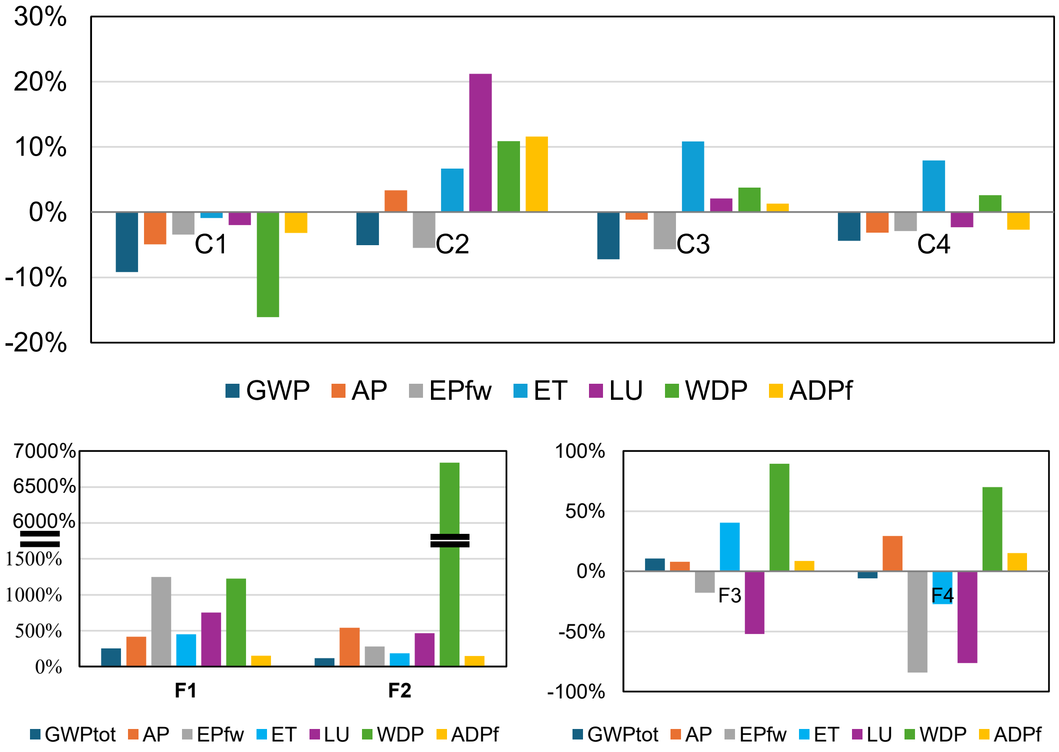

Figure C - 2: Environmental impact variation from BASE to CIR formulations for (a) C1, C2, C3, C4 in El Chami et al. (2023) and from BAU to CE formulations for (b) FC1 and F2, and (c) F3 and F4 in this work.

Table C - 17: N-P-K composition of the four assessed fertilisers in El Chami et al. (2023).

Fertiliser	**BASE**	**CIR**
TIMATECH	5 - 7 - 16	5 - 6 - 12
EUROCOD	18 - 5 - 8	16 - 5 - 6
PRIME	8 - 18 - 0	6 - 16 - 0
MAGNIFIQUE	14 - 7 - 12	13 - 7 - 10

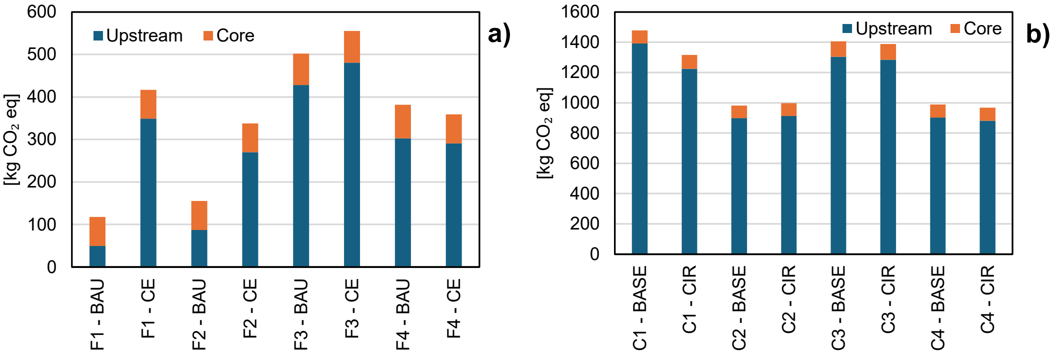

Figure C - 3: Upstream and core contributions to GWPtot for fertilisers described in this work (a) and in El Chami et al. (2023) (b).

Table C - 18: Data from official websites of main raw materials and chemicals suppliers in BAU and CE scenarios and for a hypothetical P recovery plant.

Egyptian phosphorite	The Egyptian phosphorite supplier declares its commitment to social responsibility aspects (including the improvement of workers' rights and working conditions), to effective corporate management system and to compliance with requirements from the Egyptian labour laws, the UN Declaration on Human Rights and the ILO conventions. Moreover, the supplier declares to reject illegal, forced or compulsory child labour, to provide safe and healthy working conditions and to prevent harms to people or private properties. Finally, the supplier affirms to support freedom of association and the right to collective negotiation, to reject all forms of discrimination and to maintain fair working hours and an adequate compensation.
H_2SO_4	The H_2SO_4 supplier is certified for its management system, in accordance with SA8000® standard, and for its own health and work safety management system, in accordance with ISO 45001:2018 standard.
HCl	The HCl supplier considers the responsibility on environmental protection, safety and prevention of major accidents and compliance with quality requirements of its services and products as strategic tools to achieve its objectives. The supplier is certified for its health and work safety management system, in accordance with ISO 45001:2018 standard.
$Ca(OH)_2$	The $Ca(OH)_2$ supplier is certified for its health and work safety management system, in accordance with ISO 45001:2018 standard.
LG-MgO	The LG-MgO is supplied by the private mining company MAGNESITAS NAVARRAS (Navarre, Spain). The company implemented a strong Corporate Social Responsibility policy (CSR Policy and Certifications - MAGNESITAS NAVARRAS). This policy is based on a company voluntary commitment to go beyond strict compliance with regulations, when integrating the needs of its stakeholders, and to focus on prevention, mitigation, restoration and improvement, when managing its environmental impacts. To

	implement this policy, the company establishes an annual CSR action plan with objectives and measurement indicators in the social, environmental and corporate governance areas. MAGNESITAS NAVARRAS has a certification for its occupational health and safety management system, in accordance with ISO 45001:2018 standard.
P recovery plant	For the supposed P recovery plant, the specific data for the chemical sector reported in Federchimica report (Role and Challenges of the Italian Chemical Industry) were applied. According to INAIL (Italian national institute for insurance against work accidents), the frequency of accidents in the chemical industry is half of the industrial average and the incidence of occupational diseases is the lowest in Italy.

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APPENDIX D

Appendix D

This chapter is extensively based on the following publication, under review as of January 2026:
Spotorno S., D'Agostino G., Casazza A., Perego P., Gallo M., Gagliano E., Del Borghi A. (2026*). Life Cycle Assessment and Circularity evaluation of sustainable olive pomace valorization through innovative High-Pressure High-Temperature Extraction* (*Sustainable Materials and Technologies*)

Appendix D

This appendix is referred to Chapter 7.

Table D - 1: Life Cycle Inventory for Lab-scale and simulated Pilot-scale processes. Freeze drying and spray dry process are performed for the production of the powdered biopesticide (Biopesticide A) while refrigerated storage is attributed to the liquid-form biopesticide (Biopesticide B)

			Lab-scale	Pilot-scale
Phase	**Category**	**U.M.**	**Quantity**	**Quantity**
DRYING				
	Energy consumption	kWh	56.26	0.46
HPHTE				
	Water	kg	3.15	3.15
	Ethanol	kg	2.50	2.50
	Nitrogen	l	21.00	21.00
	Energy consumption	kWh	11.11	1.12
CONV EXTRACTION				
	Water	kg	3.15	3.15
	Ethanol	kg	2.50	2.50
	Nitrogen	l	-	-
	Energy consumption	kWh	0.00	94.00
CENTRIFUGATION				
	Energy consumption	kWh	0.97	0.97
FILTRATION				
	Energy consumption	kWh	0.50	0.50
EVAPORATION (for the extract)				
	Tap water	l	839.98	839.98
	Energy consumption	kW	29.40	1.75
	Wastewater	l	839.98	839.98
FREEZE-DRYING				
	Energy consumption	kWh	282.23	-
SPRAY DRY				
	Energy consumption	kWh	-	0.00

REFRIGERATED STORAGE				
	Energy consumption	kWh	24.25	7.72
Waste disposal				
	Waste (wet residue)	kg	0.94	0.94

Table D - 2: LCIA results for all impact categories and unit operations for Biopesticide A - HPHTE - Lab-scale

Impact category [U.M.]	Total	OP Transport	DRYING	HPTE
Acidification [mol H^+ eq]	0.74	0.12	0.08	0.08
Climate change [kg CO_2 eq]	186.34	45.60	19.80	10.11
Climate change - Biogenic [kg CO_2 eq]	0.38	0.02	0.05	0.01
Climate change - Fossil [kg CO_2 eq]	185.91	45.56	19.74	10.09
Climate change - Land use and LU change [kg CO_2 eq]	0.05	0.02	0.00	0.01
Ecotoxicity, freshwater [CTUe]	583.66	181.62	45.43	64.66
Ecotoxicity, freshwater - inorganics [CTUe]	516.16	139.84	44.22	47.00
Ecotoxicity, freshwater - organics [CTUe]	67.50	41.78	1.21	17.66

Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.20	0.02	0.01	0.07
Eutrophication, freshwater [kg P eq]	0.05	0.01	0.01	0.00
Eutrophication, terrestrial [mol N eq]	1.46	0.24	0.14	0.31
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	20.00	0.87	2.77	0.91
Land use [Pt]	1587.54	206.04	107.99	653.41
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone	0.55	0.15	0.05	0.04

formation [kg NMVOC eq]				
Resource use, fossils [MJ]	2895.26	585.30	327.94	148.36
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	76.81	3.54	10.05	3.26
Non renewable, fossil [MJ]	2534.81	571.28	277.75	132.22
Non-renewable, nuclear [MJ]	360.46	14.02	50.19	16.13
Non-renewable, biomass [MJ]	0.04	0.01	0.00	0.00
Renewable, biomass [MJ]	279.72	2.89	7.11	229.91
Renewable, wind, solar, geothermal [MJ]	528.29	3.64	77.32	16.53
Renewable, water [MJ]	377.92	8.27	54.30	12.47

Impact category [U.M.]	**CENTRIFUGATION**	**FILTRATION**	**EVAPORATION**	**FREEZE-DRYING**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.00	0.05	0.40	0.00
Climate change [kg CO_2 eq]	0.34	0.21	11.02	99.27	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.00	0.04	0.25	0.00
Climate change - Fossil [kg CO_2 eq]	0.34	0.20	10.97	98.99	0.01
Climate change - Land	0.00	0.00	0.00	0.02	0.00

use and LU change [kg CO_2 eq]					
Ecotoxicity, freshwater [CTUe]	0.78	0.53	62.81	227.83	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.76	0.49	62.10	221.74	0.01
Ecotoxicity, freshwater - organics [CTUe]	0.02	0.04	0.71	6.08	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.00	0.02	0.06	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.03	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.00	0.09	0.69	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00

Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.05	0.03	1.50	13.88	0.00
Land use [Pt]	1.85	1.66	74.75	541.55	0.28
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.03	0.28	0.00
Resource use, fossils [MJ]	5.63	3.31	180.08	1644.51	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00	0.00
Water use [m3 depriv]	0.17	0.10	9.30	50.39	0.01
Non renewable, fossil [MJ]	4.77	2.81	152.98	1392.85	0.14
Non-renewable, nuclear [MJ]	0.86	0.49	27.10	251.66	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.01	0.00
Renewable, biomass [MJ]	0.12	0.17	3.87	35.65	0.00
Renewable, wind, solar, geothermal [MJ]	1.33	0.70	41.07	387.72	0.00
Renewable, water [MJ]	0.93	0.50	29.16	272.29	0.00

Table D - 3: LCIA results for all impact categories and unit operations for Biopesticide A - CONV - Lab-scale

Impact category [U.M.]	Total	OP Transport	DRYING	Total
Acidification [mol H^+ eq]	1.09	0.12	0.08	1.09
Climate change [kg CO_2 eq]	273.72	45.60	19.80	273.72
Climate change - Biogenic [kg CO_2 eq]	0.60	0.02	0.05	0.60
Climate change - Fossil [kg CO_2 eq]	273.05	45.56	19.74	273.05
Climate change - Land use and LU change [kg CO_2 eq]	0.07	0.02	0.00	0.07
Ecotoxicity, freshwater [CTUe]	784.20	181.62	45.43	784.20
Ecotoxicity, freshwater - inorganics [CTUe]	711.34	139.84	44.22	711.34
Ecotoxicity, freshwater - organics [CTUe]	72.86	41.78	1.21	72.86
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.25	0.02	0.01	0.25
Eutrophication, freshwater [kg P eq]	0.07	0.01	0.01	0.07
Eutrophication, terrestrial [mol N eq]	2.07	0.24	0.14	2.07

Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	32.22	0.87	2.77	32.22
Land use [Pt]	2064.23	206.04	107.99	2064.23
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.80	0.15	0.05	0.80
Resource use, fossils [MJ]	4342.76	585.30	327.94	4342.76
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	121.15	3.54	10.05	121.15

Non renewable, fossil [MJ]	3760.83	571.28	277.75	3760.83
Non-renewable, nuclear [MJ]	581.93	14.02	50.19	581.93
Non-renewable, biomass [MJ]	0.05	0.01	0.00	0.05
Renewable, biomass [MJ]	311.10	2.89	7.11	311.10
Renewable, wind, solar, geothermal [MJ]	869.58	3.64	77.32	869.58
Renewable, water [MJ]	617.59	8.27	54.30	617.59

Impact category [U.M.]	**CENTRIFUGATION**	**FILTRATION**	**EVAPORATION**	**FREEZE-DRYING**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.00	0.05	0.40	0.00
Climate change [kg CO_2 eq]	0.34	0.21	11.02	99.27	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.00	0.04	0.25	0.00
Climate change - Fossil [kg CO_2 eq]	0.34	0.20	10.97	98.99	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.02	0.00
Ecotoxicity, freshwater [CTUe]	0.78	0.53	62.81	227.83	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.76	0.49	62.10	221.74	0.01

Ecotoxicity, freshwater - organics [CTUe]	0.02	0.04	0.71	6.08	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.00	0.02	0.06	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.03	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.00	0.09	0.69	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.05	0.03	1.50	13.88	0.00
Land use [Pt]	1.85	1.66	74.75	541.55	0.28

Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.03	0.28	0.00
Resource use, fossils [MJ]	5.63	3.31	180.08	1644.51	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00	0.00
Water use [m3 depriv]	0.17	0.10	9.30	50.39	0.01
Non renewable, fossil [MJ]	4.77	2.81	152.98	1392.85	0.14
Non-renewable, nuclear [MJ]	0.86	0.49	27.10	251.66	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.01	0.00
Renewable, biomass [MJ]	0.12	0.17	3.87	35.65	0.00
Renewable, wind, solar, geothermal [MJ]	1.33	0.70	41.07	387.72	0.00
Renewable, water [MJ]	0.93	0.50	29.16	272.29	0.00

Table D - 4: LCIA results for all impact categories and unit operations for Biopesticide B - HPHTE - Lab-scale

Impact category [U.M.]	**Total**	**OP Transport**	**DRYING**	**HPTE**
Acidification [mol H^+ eq]	0.37	0.12	0.08	0.08

Climate change [kg CO_2 eq]	95.60	45.60	19.80	10.11
Climate change - Biogenic [kg CO_2 eq]	0.14	0.02	0.05	0.01
Climate change - Fossil [kg CO_2 eq]	95.42	45.56	19.74	10.09
Climate change - Land use and LU change [kg CO_2 eq]	0.04	0.02	0.00	0.01
Ecotoxicity, freshwater [CTUe]	375.41	181.62	45.43	64.66
Ecotoxicity, freshwater - inorganics [CTUe]	313.47	139.84	44.22	47.00
Ecotoxicity, freshwater - organics [CTUe]	61.94	41.78	1.21	17.66
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.14	0.02	0.01	0.07
Eutrophication, freshwater [kg P eq]	0.02	0.01	0.01	0.00
Eutrophication, terrestrial [mol N eq]	0.83	0.24	0.14	0.31
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00

Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	7.31	0.87	2.77	0.91
Land use [Pt]	1092.53	206.04	107.99	653.41
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.30	0.15	0.05	0.04
Resource use, fossils [MJ]	1392.05	585.30	327.94	148.36
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	30.75	3.54	10.05	3.26
Non renewable, fossil [MJ]	1261.64	571.28	277.75	132.22
Non-renewable, nuclear [MJ]	130.42	14.02	50.19	16.13
Non-renewable, biomass [MJ]	0.03	0.01	0.00	0.00

Renewable, biomass [MJ]	247.14	2.89	7.11	229.91
Renewable, wind, solar, geothermal [MJ]	173.89	3.64	77.32	16.53
Renewable, water [MJ]	129.03	8.27	54.30	12.47

Impact category [U.M.]	CENTRIFUGATION	FILTRATION	EVAPORATION	REF. STORAGE	Waste treatment
Acidification [mol H^+ eq]	0.00	0.00	0.05	0.03	0.00
Climate change [kg CO_2 eq]	0.34	0.21	11.02	8.53	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.00	0.04	0.02	0.00
Climate change - Fossil [kg CO_2 eq]	0.34	0.20	10.97	8.51	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	0.78	0.53	62.81	19.58	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.76	0.49	62.10	19.05	0.01
Ecotoxicity, freshwater - organics [CTUe]	0.02	0.04	0.71	0.52	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00	0.00

Eutrophication, marine [kg N eq]	0.00	0.00	0.02	0.01	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.00	0.09	0.06	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.05	0.03	1.50	1.19	0.00
Land use [Pt]	1.85	1.66	74.75	46.53	0.28
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.03	0.02	0.00

Resource use, fossils [MJ]	5.63	3.31	180.08	141.30	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00	0.00
Water use [m3 depriv]	0.17	0.10	9.30	4.33	0.01
Non renewable, fossil [MJ]	4.77	2.81	152.98	119.68	0.14
Non-renewable, nuclear [MJ]	0.86	0.49	27.10	21.62	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.00	0.00
Renewable, biomass [MJ]	0.12	0.17	3.87	3.06	0.00
Renewable, wind, solar, geothermal [MJ]	1.33	0.70	41.07	33.31	0.00
Renewable, water [MJ]	0.93	0.50	29.16	23.40	0.00

Table D - 5: LCIA results for all impact categories and unit operations for Biopesticide B - CONV - Lab-scale

Impact category [U.M.]	**Total**	**OP Transport**	**DRYING**	**CONV**
Acidification [mol H^+ eq]	0.72	0.12	0.08	0.44
Climate change [kg CO_2 eq]	182.98	45.60	19.80	97.49

Climate change - Biogenic [kg CO_2 eq]	0.37	0.02	0.05	0.24
Climate change - Fossil [kg CO_2 eq]	182.56	45.56	19.74	97.23
Climate change - Land use and LU change [kg CO_2 eq]	0.05	0.02	0.00	0.02
Ecotoxicity, freshwater [CTUe]	575.95	181.62	45.43	265.19
Ecotoxicity, freshwater - inorganics [CTUe]	508.65	139.84	44.22	242.18
Ecotoxicity, freshwater - organics [CTUe]	67.30	41.78	1.21	23.01
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.19	0.02	0.01	0.13
Eutrophication, freshwater [kg P eq]	0.05	0.01	0.01	0.03
Eutrophication, terrestrial [mol N eq]	1.44	0.24	0.14	0.91
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer	0.00	0.00	0.00	0.00

- organics [CTUh]				
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	19.53	0.87	2.77	13.13
Land use [Pt]	1569.22	206.04	107.99	1130.10
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.54	0.15	0.05	0.28
Resource use, fossils [MJ]	2839.55	585.30	327.94	1595.85
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	75.09	3.54	10.05	47.60
Non renewable, fossil [MJ]	2487.66	571.28	277.75	1358.25
Non-renewable, nuclear [MJ]	351.89	14.02	50.19	237.61
Non-renewable, biomass [MJ]	0.04	0.01	0.00	0.02
Renewable, biomass [MJ]	278.51	2.89	7.11	261.29

Renewable, wind, solar, geothermal [MJ]	515.17	3.64	77.32	357.81
Renewable, water [MJ]	368.70	8.27	54.30	252.15

Impact category [U.M.]	**CENTRIFUGATION**	**FILTRATION**	**EVAPORATION**	**REF. STORAGE**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.00	0.05	0.03	0.00
Climate change [kg CO_2 eq]	0.34	0.21	11.02	8.53	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.00	0.04	0.02	0.00
Climate change - Fossil [kg CO_2 eq]	0.34	0.20	10.97	8.51	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	0.78	0.53	62.81	19.58	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.76	0.49	62.10	19.05	0.01
Ecotoxicity, freshwater - organics [CTUe]	0.02	0.04	0.71	0.52	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.00	0.02	0.01	0.00

Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.00	0.09	0.06	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.05	0.03	1.50	1.19	0.00
Land use [Pt]	1.85	1.66	74.75	46.53	0.28
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.03	0.02	0.00
Resource use, fossils [MJ]	5.63	3.31	180.08	141.30	0.14
Resource use, minerals and	0.00	0.00	0.00	0.00	0.00

metals [kg Sb eq]					
Water use [m3 depriv]	0.17	0.10	9.30	4.33	0.01
Non renewable, fossil [MJ]	4.77	2.81	152.98	119.68	0.14
Non-renewable, nuclear [MJ]	0.86	0.49	27.10	21.62	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.00	0.00
Renewable, biomass [MJ]	0.12	0.17	3.87	3.06	0.00
Renewable, wind, solar, geothermal [MJ]	1.33	0.70	41.07	33.31	0.00
Renewable, water [MJ]	0.93	0.50	29.16	23.40	0.00

Table D - 6: LCIA results for all impact categories and unit operations for Biopesticide A - HPHTE - Pilot-scale

Impact category [U.M.]	**Total**	**DRYING**	**HPTE**	**CENTRIFUGATION**
Acidification [mol H^+ eq]	0.09	0.00	0.07	0.00
Climate change [kg CO_2 eq]	11.32	0.16	6.60	0.34
Climate change - Biogenic [kg CO_2 eq]	0.03	0.00	0.01	0.00

Climate change - Fossil [kg CO_2 eq]	11.29	0.16	6.59	0.34
Climate change - Land use and LU change [kg CO_2 eq]	0.01	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	105.01	0.37	56.59	0.78
Ecotoxicity, freshwater - inorganics [CTUe]	87.22	0.36	39.15	0.76
Ecotoxicity, freshwater - organics [CTUe]	17.79	0.01	17.44	0.02
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.09	0.00	0.07	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00
Eutrophication, terrestrial [mol N eq]	0.33	0.00	0.28	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00

Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	1.03	0.02	0.42	0.05
Land use [Pt]	675.46	0.89	634.24	1.86
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.04	0.00	0.03	0.00
Resource use, fossils [MJ]	165.94	2.70	90.13	5.65
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	7.58	0.08	1.47	0.17
Non renewable, fossil [MJ]	147.61	2.28	82.91	4.79
Non-renewable, nuclear [MJ]	18.33	0.41	7.22	0.86
Non-renewable, biomass [MJ]	0.01	0.00	0.00	0.00
Renewable, biomass [MJ]	230.36	0.06	228.65	0.12
Renewable, wind, solar, geothermal [MJ]	19.17	0.64	2.80	1.33

Renewable, water [MJ]	14.65	0.45	2.83	0.94

Impact category [U.M.]	**FILTRATION**	**EVAPORATION**	**FREEZE-DRYING**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.01	0.01	0.00
Climate change [kg CO_2 eq]	0.21	1.29	2.72	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.01	0.01	0.00
Climate change - Fossil [kg CO_2 eq]	0.20	1.28	2.71	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	0.53	40.49	6.23	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.49	40.39	6.07	0.01
Ecotoxicity, freshwater - organics [CTUe]	0.04	0.11	0.17	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.02	0.00	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00

Eutrophication, terrestrial [mol N eq]	0.00	0.02	0.02	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.03	0.14	0.38	0.00
Land use [Pt]	1.66	21.72	14.81	0.28
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.01	0.00
Resource use, fossils [MJ]	3.31	19.03	44.98	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00

Water use [m3 depriv]	0.10	4.36	1.38	0.01
Non renewable, fossil [MJ]	2.81	16.58	38.10	0.14
Non-renewable, nuclear [MJ]	0.49	2.45	6.88	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.00
Renewable, biomass [MJ]	0.17	0.38	0.98	0.00
Renewable, wind, solar, geothermal [MJ]	0.70	3.10	10.61	0.00
Renewable, water [MJ]	0.50	2.49	7.45	0.00

Table D - 7: LCIA results for all impact categories and unit operations for Biopesticide A - CONV - Pilot-scale

Impact category [U.M.]	Total	DRYING	CONV	CENTRIFUGATION
Acidification [mol H^+ eq]	0.22	0.00	0.20	0.00
Climate change [kg CO_2 eq]	43.99	0.16	39.27	0.34
Climate change - Biogenic [kg CO_2 eq]	0.11	0.00	0.09	0.00
Climate change - Fossil [kg CO_2 eq]	43.87	0.16	39.17	0.34
Climate change - Land use and LU change [kg CO_2 eq]	0.01	0.00	0.01	0.00
Ecotoxicity, freshwater [CTUe]	179.99	0.37	131.56	0.78
Ecotoxicity, freshwater - inorganics [CTUe]	160.20	0.36	112.12	0.76
Ecotoxicity, freshwater - organics [CTUe]	19.79	0.01	19.45	0.02
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.11	0.00	0.09	0.00
Eutrophication, freshwater [kg P eq]	0.01	0.00	0.01	0.00
Eutrophication, terrestrial [mol N eq]	0.55	0.00	0.51	0.00

Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	5.60	0.02	4.99	0.05
Land use [Pt]	853.68	0.89	812.45	1.86
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.13	0.00	0.12	0.00
Resource use, fossils [MJ]	707.13	2.70	631.32	5.65
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	24.16	0.08	18.05	0.17

Non renewable, fossil [MJ]	605.98	2.28	541.28	4.79
Non-renewable, nuclear [MJ]	101.15	0.41	90.04	0.86
Non-renewable, biomass [MJ]	0.01	0.00	0.01	0.00
Renewable, biomass [MJ]	242.09	0.06	240.38	0.12
Renewable, wind, solar, geothermal [MJ]	146.77	0.64	130.40	1.33
Renewable, water [MJ]	104.26	0.45	92.44	0.94

Impact category [U.M.]	**FILTRATION**	**EVAPORATION**	**FREEZE-DRYING**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.01	0.01	0.00
Climate change [kg CO_2 eq]	0.21	1.29	2.72	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.01	0.01	0.00
Climate change - Fossil [kg CO_2 eq]	0.20	1.28	2.71	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	0.53	40.49	6.23	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.49	40.39	6.07	0.01

Ecotoxicity, freshwater - organics [CTUe]	0.04	0.11	0.17	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.02	0.00	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.02	0.02	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.03	0.14	0.38	0.00
Land use [Pt]	1.66	21.72	14.81	0.28

Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.01	0.00
Resource use, fossils [MJ]	3.31	19.03	44.98	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	0.10	4.36	1.38	0.01
Non renewable, fossil [MJ]	2.81	16.58	38.10	0.14
Non-renewable, nuclear [MJ]	0.49	2.45	6.88	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.00
Renewable, biomass [MJ]	0.17	0.38	0.98	0.00
Renewable, wind, solar, geothermal [MJ]	0.70	3.10	10.61	0.00
Renewable, water [MJ]	0.50	2.49	7.45	0.00

Table D - 8: LCIA results for all impact categories and unit operations for Biopesticide B - HPHTE - Pilot-scale

Impact category [U.M.]	Total	DRYING	HPHTE	CENTRIFUGATION
Acidification [mol H^+ eq]	0.08	0.00	0.07	0.00
Climate change [kg CO_2 eq]	8.74	0.16	6.60	0.34
Climate change - Biogenic [kg CO_2 eq]	0.02	0.00	0.01	0.00
Climate change - Fossil [kg CO_2 eq]	8.71	0.16	6.59	0.34
Climate change - Land use and LU change [kg CO_2 eq]	0.01	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	99.09	0.37	56.59	0.78
Ecotoxicity, freshwater - inorganics [CTUe]	81.46	0.36	39.15	0.76
Ecotoxicity, freshwater - organics [CTUe]	17.63	0.01	17.44	0.02
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.09	0.00	0.07	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00

Eutrophication, terrestrial [mol N eq]	0.31	0.00	0.28	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.67	0.02	0.42	0.05
Land use [Pt]	661.38	0.89	634.24	1.86
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.04	0.00	0.03	0.00
Resource use, fossils [MJ]	123.18	2.70	90.13	5.65
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00

Water use [m3 depriv]	6.27	0.08	1.47	0.17
Non renewable, fossil [MJ]	111.39	2.28	82.91	4.79
Non-renewable, nuclear [MJ]	11.79	0.41	7.22	0.86
Non-renewable, biomass [MJ]	0.01	0.00	0.00	0.00
Renewable, biomass [MJ]	229.43	0.06	228.65	0.12
Renewable, wind, solar, geothermal [MJ]	9.09	0.64	2.80	1.33
Renewable, water [MJ]	7.57	0.45	2.83	0.94

Impact category [U.M.]	**FILTRATION**	**EVAPORATION**	**FREEZE-DRYING**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.01	0.00	0.00
Climate change [kg CO_2 eq]	0.21	1.29	0.13	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.01	0.00	0.00
Climate change - Fossil [kg CO_2 eq]	0.20	1.28	0.13	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	0.53	40.49	0.31	0.01
Ecotoxicity, freshwater -	0.49	40.39	0.30	0.01

inorganics [CTUe]				
Ecotoxicity, freshwater - organics [CTUe]	0.04	0.11	0.01	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.02	0.00	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.02	0.00	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.03	0.14	0.02	0.00

Land use [Pt]	1.66	21.72	0.73	0.28
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.00	0.00
Resource use, fossils [MJ]	3.31	19.03	2.21	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	0.10	4.36	0.07	0.01
Non renewable, fossil [MJ]	2.81	16.58	1.88	0.14
Non-renewable, nuclear [MJ]	0.49	2.45	0.34	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.00
Renewable, biomass [MJ]	0.17	0.38	0.05	0.00
Renewable, wind, solar, geothermal [MJ]	0.70	3.10	0.52	0.00
Renewable, water [MJ]	0.50	2.49	0.37	0.00

Table D - 9: LCIA results for all impact categories and unit operations for Biopesticide B - HPHTE - Pilot-scale

Impact category [U.M.]	Total	DRYING	CONV	CENTRIFUGATION
Acidification [mol H^+ eq]	0.21	0.00	0.20	0.00
Climate change [kg CO_2 eq]	41.41	0.16	39.27	0.34
Climate change - Biogenic [kg CO_2 eq]	0.11	0.00	0.09	0.00
Climate change - Fossil [kg CO_2 eq]	41.29	0.16	39.17	0.34
Climate change - Land use and LU change [kg CO_2 eq]	0.01	0.00	0.01	0.00
Ecotoxicity, freshwater [CTUe]	174.06	0.37	131.56	0.78
Ecotoxicity, freshwater - inorganics [CTUe]	154.43	0.36	112.12	0.76
Ecotoxicity, freshwater - organics [CTUe]	19.63	0.01	19.45	0.02
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.11	0.00	0.09	0.00
Eutrophication, freshwater [kg P eq]	0.01	0.00	0.01	0.00
Eutrophication, terrestrial [mol N eq]	0.54	0.00	0.51	0.00

Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	5.24	0.02	4.99	0.05
Land use [Pt]	839.60	0.89	812.45	1.86
Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.13	0.00	0.12	0.00
Resource use, fossils [MJ]	664.37	2.70	631.32	5.65
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	22.85	0.08	18.05	0.17

Non renewable, fossil [MJ]	569.76	2.28	541.28	4.79
Non-renewable, nuclear [MJ]	94.61	0.41	90.04	0.86
Non-renewable, biomass [MJ]	0.01	0.00	0.01	0.00
Renewable, biomass [MJ]	241.16	0.06	240.38	0.12
Renewable, wind, solar, geothermal [MJ]	136.68	0.64	130.40	1.33
Renewable, water [MJ]	97.18	0.45	92.44	0.94

Impact category [U.M.]	**FILTRATION**	**EVAPORATION**	**FREEZE-DRYING**	**Waste treatment**
Acidification [mol H^+ eq]	0.00	0.01	0.00	0.00
Climate change [kg CO_2 eq]	0.21	1.29	0.13	0.01
Climate change - Biogenic [kg CO_2 eq]	0.00	0.01	0.00	0.00
Climate change - Fossil [kg CO_2 eq]	0.20	1.28	0.13	0.01
Climate change - Land use and LU change [kg CO_2 eq]	0.00	0.00	0.00	0.00
Ecotoxicity, freshwater [CTUe]	0.53	40.49	0.31	0.01
Ecotoxicity, freshwater - inorganics [CTUe]	0.49	40.39	0.30	0.01

Ecotoxicity, freshwater - organics [CTUe]	0.04	0.11	0.01	0.00
Particulate matter [disease inc]	0.00	0.00	0.00	0.00
Eutrophication, marine [kg N eq]	0.00	0.02	0.00	0.00
Eutrophication, freshwater [kg P eq]	0.00	0.00	0.00	0.00
Eutrophication, terrestrial [mol N eq]	0.00	0.02	0.00	0.00
Human toxicity, cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - inorganics [CTUh]	0.00	0.00	0.00	0.00
Human toxicity, non-cancer - organics [CTUh]	0.00	0.00	0.00	0.00
Ionising radiation [kBq U-235 eq]	0.03	0.14	0.02	0.00
Land use [Pt]	1.66	21.72	0.73	0.28

Ozone depletion [kg CFC11 eq]	0.00	0.00	0.00	0.00
Photochemical ozone formation [kg NMVOC eq]	0.00	0.00	0.00	0.00
Resource use, fossils [MJ]	3.31	19.03	2.21	0.14
Resource use, minerals and metals [kg Sb eq]	0.00	0.00	0.00	0.00
Water use [m3 depriv]	0.10	4.36	0.07	0.01
Non renewable, fossil [MJ]	2.81	16.58	1.88	0.14
Non-renewable, nuclear [MJ]	0.49	2.45	0.34	0.00
Non-renewable, biomass [MJ]	0.00	0.00	0.00	0.00
Renewable, biomass [MJ]	0.17	0.38	0.05	0.00
Renewable, wind, solar, geothermal [MJ]	0.70	3.10	0.52	0.00
Renewable, water [MJ]	0.50	2.49	0.37	0.00