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Reusing drainage water and substrate to improve the environmental and economic performance of Mediterranean greenhouse cropping

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Highlights

- Reusing drainage water can save 20% water and fertilisers
- Reusing drainage water can abate over 50% freshwater and marine eutrophication
- Reuse strategies are viable (net present value over 20 years > € 400000)
- Avoiding substrate landfilling increases the profitability of reuse technologies
- Cascade cropping and closed-loop fertigation are eco-efficient reuse technologies

Abstract

The objective of this study is to provide decision makers and policy makers with adequate information to support the diffusion of reuse strategies in Mediterranean greenhouses. Sixteen alternative scenarios are compared through eco-efficiency analysis, combining four technologies to manage drainage water (open-loop fertigation vs. wastewater treatment plant vs. cascade cropping vs. closed-loop fertigation) with two substrate materials and two substrate management options at end-of-life. System differences are modelled through detailed primary data, collected and validated via a multi-step process. Results show that cascade cropping and closed-loop fertigation have, respectively, the highest and second-highest eco-efficiency, with respect to their ability to reduce freshwater eutrophication (up to -6,63 kg P) and marine eutrophication (up to -47.1 kg P eq), while generating profits for the farmer. Selecting a biodegradable substrate and reusing it on farm can increase greenhouse profitability by 20%. This article is a new contribution to the literature by (i) supporting the improvement and harmonisation of eco-efficiency analysis in the agricultural sector; (ii) providing a comprehensive comparative assessment that is missing from the published literature; (iii) giving special emphasis to data and the data collection process, to provide input to further research; (iv) by generating lessons learnt of practical usefulness for reducing uncertainty in decision making and policy making; (v) by delivering policy recommendations to address key barriers to the diffusion of eco-efficient greenhouse cropping. The involvement of local and multidisciplinary stakeholders is required to improve the methodological approach and the acceptability of the proposed solution, especially in case of trade-offs among the different impact domains, and to identify and prioritise tailored interventions on the conditions and stakeholder needs.

Keywords: Impact assessment; horticultural substrate; closed-loop fertigation; cascade cropping; wastewater treatment; future cash flows

Nomenclature

Scenarios-subscenarios

BAU: business as usual

WWTP: wastewater treatment plant

CSC: cascade cropping

CLS: closed-loop fertigation

sw-o: stone wool, ordinary management of exhausted substrate (landfill)

sw-r: stone wool, reuse of exhausted substrate (recycling)

cp-o: coir pith, ordinary management of exhausted substrate (land spreading)

cp-r: stone wool, reuse of exhausted substrate (composting)

Life cycle assessment

AC: Acidification

CC: Climate change

FET: Freshwater toxicity

FE: Freshwater eutrophication

HTC: Human toxicity cancer

HTnC: Human toxicity non-cancer

LU: Land use

ME: Marine eutrophication

MFR: Mineral, fossil and renewable resource depletion

OD: Ozone depletion

POF: Photochemical ozone formation

TE: Terrestrial eutrophication

WRD: Water resource depletion

Life cycle costing

TCOP: Total costs of production

NPV: Net present value

PI: Profitability index

1. Introduction

1.1. Research motivation and objective

Greenhouse horticulture faces the challenge of how to meet the growing demand for fresh vegetables, while reducing the impacts on the environment and human health and ensuring agricultural viability (Euromonitor International, 2018; Pretty and Bharucha, 2014; Thompson et al., 2020). Addressing this challenge is crucial to the achievement of Sustainable Development Goals while meeting planetary boundaries (Rockström et al., 2009; Sutton et al., 2021; UN, 2015), and to enable sustainable healthy diets, including greater consumption of fresh vegetables (FAO and WHO, 2019; Mason-D'Croz et al., 2019; Yin et al., 2020). In the European Union, addressing that challenge would contribute to achieving the zero pollution ambitions of the European Commission's Green Deal (European Commission, 2020a) and more specific objectives of the Farm to Fork Strategy (European Commission, 2020b) and the Circular Economy Action Plan (European Commission, 2020c).

In Europe, key greenhouse vegetables are produced year-round in unheated greenhouses in Mediterranean countries, using soilless systems (i.e. on cultivation substrates) (Incrocci et al., 2020; Massa et al., 2020). Compared to soil cropping, soilless cropping has a greater efficiency of fertiliser and water use (Savvas et al., 2013; Savvas and Gruda, 2018), given the better water retention properties of substrates compared to soil (Nikolaou et al., 2019; Putra and Yuliando, 2015). However, in Mediterranean countries soilless cropping generates serious environmental impacts, due the great diffusion of open-loop fertigation, where the excess nutrient solution after meeting crop needs (drainage water) is discharged to the ground (Grewal et al., 2011; Thompson et al., 2020). Adopting strategies to reuse drainage water can save up to 40% irrigation water and up to 50% emissions from fertilisers, without significantly affecting crop productivity (Grewal et al., 2011; Komosa et al., 2011; Meric et al., 2011). Besides drainage water, reuse strategies should consider at least the management of the cultivation substrate, a key element of soilless cropping, (Barrett et al., 2016; EIP-AGRI, 2019a) and the economic feasibility of the proposed interventions, given the cost-related barriers that have prevented environmental sustainability improvements in commercial Mediterranean greenhouses (EIP-AGRI, 2019b; Juntti and Downward, 2017). Reusable substrate should be promoted that offers a good compromise between technological and environmental performance, and purchase and end-of-life costs for the farmer (Barrett et al., 2016; Gruda, 2019; QUANTIS, 2012; Savvas and Gruda, 2018). The proposed interventions should consider incremental technologies, i.e. that can be modulated based on context-specific factors, including the ease of access to loans for the farmer (Norman and Verganti, 2014; Pearce et al., 2018).

Against that background, the overarching objective of this study is to show the potential of alternative reuse technologies, to improve the environmental and economic performance of soilless greenhouse cultivation in a Mediterranean context. More specifically, this objective is achieved by addressing two research questions (RQ) that, to the best of authors' knowledge, are still unanswered:

RQ1: "What are the environmental-economic trade-offs of incremental technologies to enable the reuse of drainage water and cultivation substrate in commercial Mediterranean greenhouses?"

RQ2: "What are the best value-for-money technologies, readily available on the market, that can enable the diffusion of reuse strategies across Mediterranean greenhouses in a timely manner?".

Addressing those research questions requires a life cycle approach, as different types and quantities of materials, with different useful lives, are needed for distinct technologies and their relative maintenance (Guinée et al., 2011; Heijungs et al., 2009; Rajagopal et al., 2017). Different methods exist to consider the production inputs and outputs throughout the life cycle of a product, e.g. life cycle assessment, life cycle costing, material flow analysis, environmentally-extended input output analysis or cost-benefit analysis (Finnveden and Moberg, 2005; Hoogmartens et al., 2014). Method selection depends on the aims and scope of the study (for example material flow analysis does not include an impact assessment, environmentally-extended input-output analysis is an economy-wide assessment that can be carried out at the country or higher level (Reimann et al., 2010)), which includes identifying the way how to deal with environmental-economic trade-offs, as well (Hamilton et al., 2015; Huguet Ferran et al., 2018), e.g. via multicriteria decision analysis, data envelopment analysis or eco-efficiency analysis (Cook et al., 2014; Rüdenauer et al., 2005; Stewart, 1996).

The adopted research method is Eco-efficiency analysis (EE), based on the combination of Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) at the farm level. LCA (ISO 14040:2006; 14044: 2006) and LCC (a standard exists for the building sector, ISO 15686-5:2017) are widely applied, individually or in combination, for the evaluation of alternative vegetable production technologies (Cellura et al., 2012; Peña and Rovira-Val, 2020; Sanyé-Mengual et al., 2015; Tamburini et al., 2015; Testa et al., 2014a; Torrellas et al., 2012a). LCA and LCC are suitable for micro-level assessments and well accepted and known by stakeholders (Reimann et al., 2010).

Data (2014-2018) refer to a typical farm central Italy (Tuscany), the production system and technology of which are reasonably representative of the Mediterranean context (Almeida et al., 2014;

Cellura et al., 2012; Testa et al., 2014b). Tomatoes are just one of the many crop species that are suitable for soilless production. Like similar research (Hollingsworth et al., 2020), this study uses tomatoes as a reference crop because it is the most commonly grown, and the highest-value added, greenhouse crop in Mediterranean Europe (De Cicco, 2019; European Commission, 2020d). Tomatoes are the most widely consumed horticultural products in the world (OECD, 2017) and are expected to be a central crop in changing diets (European Commission, 2020d).

1.2. Contribution to the relevant literature

Eco-efficiency analysis (ISO 14045:2012) (Schmidheiny, 1992; WBCSD, 2005) generates evidence about the best value-for-money interventions to improve the sustainability of production systems (Caiado et al., 2017; Miah et al., 2017). Such evidence can be used to target policy support and guide decision making (Zhang et al., 2019; Zhen et al., 2020), by linking environmental impact indicators calculated via LCA with economic impact indicators (Huppes and Ishikawa, 2005a, 2005b; Rüdenauer et al., 2005; Saling et al., 2002). The EE standard does not identify a specific method for the economic impact assessment; however, researchers agree on the use of LCC, subject to internal consistency among methodological choices for LCA and LCC (e.g. system boundaries and functional units) (Kirchherr et al., 2017; Koskela and Vehmas, 2012; Saling, 2016; Todorovic et al., 2016).

There are a series of EE variants, mainly differing for: (i) the use of absolute vs. relative values for the environmental and economic impact indicators; (ii) the use of a single score LCA (after normalisation and weighting) vs. individual results per impact category; (iii) the use of current vs. discounted economic values; (iv) the way how a relationship is created between the outputs LCC and LCA, i.e. by calculating a ratio, by adding them or by plotting them on a two-way graph (Huppes and Ishikawa, 2005b, 2005a; Koskela and Vehmas, 2012; UNEP/SETAC, 2008; Zhang et al., 2019).

In greenhouse horticulture, EE stands on the shoulders of the large LCA literature, which is especially developed for tomatoes (Torres Pineda et al., 2020). Over the last 20 years, LCA studies have assessed different production systems to identify the opportunities for sustainable greenhouse vegetable production (Perrin et al., 2014), such as infrastructures suitable to different climates (Torrellas et al., 2012b), renewable energy (Maaoui et al., 2021), systems to improve energy efficiency (Antón et al., 2012; Dorais et al., 2014) and to close the fertigation loop (Antón et al., 2005; Page et al., 2012), cascade cropping (Muñoz et al., 2017), different lighting systems (Zhang et al., 2017), to cite a few. LCAs are available to support substrate selection as well (Dorr et al., 2017; Vinci and Rapa, 2019). LCC studies are available for different greenhouse crops and production methods (Banaeian et al., 2011; Mohamad et al., 2018; Mohammadi and Omid, 2010; Testa et al., 2014a) and are often combined with LCAs (Peña and Rovira-Val, 2020), especially for the assessment of innovative production systems (Sanyé-Mengual et al., 2017). The combination of LCA and LCC into EE has got growing attention by agricultural research (Suzigan et al., 2020), with examples from different sectors, e.g. dairy products (Forleo et al., 2018a; Skrydstrup et al., 2020), cereals (Babu et al., 2020; Chancharoonpong et al., 2021; Kumar et al., 2021; Saber et al., 2021; Todorović et al., 2018), energy crops (Forleo et al., 2018b; Kochaphum et al., 2015), orchard fruit (Kim et al., 2020; Mouron et al., 2006; Müller et al., 2015), as well as horticultural crops (Mohammadzadeh et al., 2018; Sanyé-Mengual et al., 2018). In greenhouse horticulture, EE have focused on conventional vs. organic production methods (Zhen et al., 2020), crop selection in rooftop production systems (Rufí-Salís et al., 2020a), different lighting systems (Pennisi et al., 2019).

This article is a new contribution to agricultural EE by supporting method development and by showing new empirical findings. First, the article supports the need for method harmonization by presenting an approach to EE that uses discounted economic values based on LCC and plots relative changes in all LCA outputs and the economic indicator on a graph. Second, the article adds evidence to the published literature by (i) reporting a structured data collection process, which results in a detailed data source for further research; and (ii) delivering a comprehensive assessment of the potential environmental-economic impacts of reuse technologies for drainage water and the substrate that can be promptly adopted by farmers. Additionally, the article will contribute to the debate on ecological transitions of agri-food systems, by providing evidence about the eco-efficiency of incremental innovation in commercial greenhouses.

2. Research methods

The goal of this study is to compare the cradle-to-gate eco-efficiency (LCA + LCC) of three alternative and incremental reuse technologies for greenhouse tomato production in soilless culture against the ordinary production system. The assessment is based on real-world data collected in central Italy that are representative for the sector in Mediterranean Europe, and targets researchers and decision-makers in agribusiness and agricultural policy, who need to identify, assess, and prioritise sustainability interventions, as well as to develop long-term strategies. The functional unit is the occupation of 1 ha of greenhouse area for producing soilless tomatoes for 1 year. An area-based functional unit is selected against a mass-based one for two reasons, i.e. the focus of the study on management decisions and the fact that the modelled reuse technologies do not significantly affect greenhouse productivity (Charles et al., 2006). No allocation is considered since there is only one marketable product. The system under study is defined by a series of inputs and outputs occurring at different life cycle stages (Figure 1).

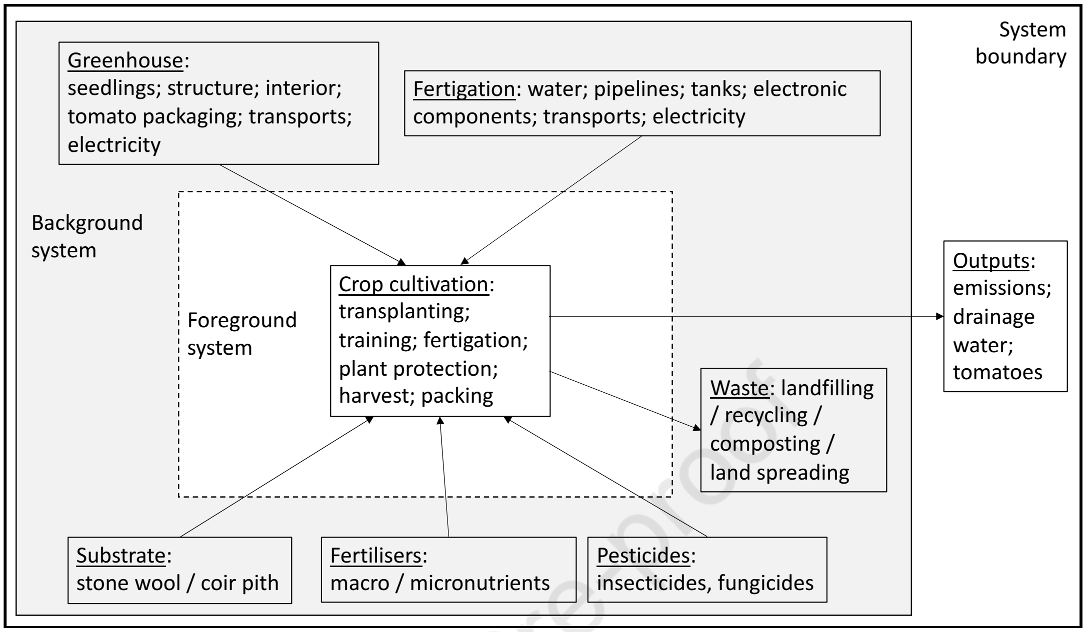

Figure 1. System boundaries. Source: Authors' own elaboration.

The foreground system is defined by the use phase, i.e. the set-up and management of soilless cultivation by farm labour, including plant training, seedling transplanting, replacement of exhausted with new substrate, use of water, electricity, fertilisers and pesticides, as well as harvesting and packing marketable tomatoes. The background system includes the production, manufacturing, assembly, maintenance, dismantling and end-of life of all the materials, resources and energy used. The outputs are emissions to the environment, drainage water and marketable tomatoes.

In soilless systems, the applied nutrient solution exceeds crop needs by about 30 %, to ensure proper crop development and optimal yield (Sonneveld & Voogt, 2009)). When this surplus (drainage water) is not properly managed, like in open-loop soilless culture, crop nutrients leach to the ground and generate environmental problems (Incrocci et al., 2020; Kläring, 2001). To avoid those problems, technology is readily available for farm uptake that allows to collect the surplus solution and to reuse it for different purposes, as follows:

1. Recovering water for indirect uses via treatment in a municipal wastewater treatment plant (EIP-AGRI, 2019c);
2. Recovering water and nutrients for the fertigation of other soil-grown greenhouse crops or for growing more salt resistant soilless crops on the same farm (cascade cropping) (Elvanidi et al., 2020; García-Caparrós et al., 2018);

3. Recirculating drainage water on the same crop (closed-loop fertigation) (Savvas and Gruda, 2018).

Those strategies are not mutually exclusive and can be considered by the decision-maker for incremental changes.

To achieve the goal of the study four scenarios are built to represent open-loop fertigation and the three reuse strategies above.

To extend the usefulness of the study to the Mediterranean context, the assessment considers two widely used substrates, i.e. rockwool, as observed in the case study, and coir pith (Massa et al., 2020) and two different end-life treatments for each substrate. This is done by developing four subscenarios.

Key assumptions for each scenario and subscenario are described below.

2.1. Scenarios and subscenarios

BAU - business as usual scenario. This scenario shows what if nothing changes compared to the real-world situation observed in the case study. Key assumptions:

- Direct emissions from fertilisers and pesticides: no emissions to soil are considered, as it is assumed that all soluble crop nutrients leach to water bodies and that pesticides are sprayed with closed windows (cf. Maaoui et al., 2021), so emissions to air can be calculated considering a drift fraction of 5% applied quantity of active ingredients (Juraske et al., 2007).
- The farmer keeps the greenhouse in optimal operating conditions, by guaranteeing the following material useful lives: concrete and metals, 20 years; fertigation/sterilisation control units, 10 years; floor mulching and plastic tanks, 5 years; the rest of plastics, 3 years, but for raffia thread, clips and wedges for plant training that are replaced twice per year; pollination hives (cardboard) are replaced twice per year; substrate, 2 years (Torrellas et al., 2012a);
- Waste is collected by the local waste company and sorted, based on quality, for proper allocation among the treatment or recycling facilities. Waste stage is modelled, based on the cut-off method (Ekvall and Tillman, 1997), as follows: 50% concrete, metals, plastics, cardboard, electric and electronic materials are recycled and 50% landfilled.

Largely, those assumptions hold for the three reuse scenarios, as well. In the next subsections, key assumptions are presented just when differing from BAU.

WWTP – drainage water treated in a municipal wastewater treatment plant scenario. This scenario shows what if BAU is upgraded with the infrastructure to prevent drainage water leaching, by collecting and delivering it to a municipal wastewater treatment plant. A series of emissions to air and water are associated with wastewater treatment before releasing water that does not cause further pollution of water sources (Guven et al., 2018). Key assumptions:

- Tomato yield is equal to BAU;
- The whole yearly volume of drainage water is collected and treated, so there are no direct emissions from fertilisers to the water compartment;
- Drainage water is delivered to the closest municipal wastewater treatment plant once per week, by completely emptying the collection tank so that tank plastic materials can be easily replaced at the end of their useful lives.

<u>CSC – cascade cropping scenario</u>. This scenario reproduces WWTP showing what if the collected drainage water is entirely used for the fertigation of a soil-grown greenhouse crop on farm (instead of delivering it to a wastewater treatment plant). Key assumptions:

- The whole yearly volume of drainage water is recycled on a second crop, so there are no direct emissions from fertilisers to the water compartment (Ekvall and Tillman, 1997).
- The second cultivation occurs on farm on a neighbouring greenhouse;
- The second crop is melon, which is suitable for cascade cropping systems and for the climate conditions in the case study area and similar greenhouse systems;
- Drainage water is suitable for the fertigation of melon under ordinary conditions in the case study area and in similar greenhouse systems (duration of crop cycle = 120 days; 1 cycle per year; yield 30 t/ha) subject to accurate salt and nutrient concentration adjustments, by the farmer to ensure proper crop development (nutritional needs of melon in kg/ha/yr: N = 165; P_2O_5 = 60; K_2O = 220; CaO = 60; MgO = 40; Fe = 5; B = 5; Cu = 1; Zn = 4; Mn = 1; based on researchers' experience and expert interviews (cf. Cellura et al., 2012; Martin-Gorriz et al., 2020).

<u>CLS - closed-loop fertigation scenario</u>. This scenario shows what if BAU is upgraded with the infrastructure to recirculate the entire volume of drainage water on the same crop, subject to filtration, sterilisation and salt and nutrient concentration adjustments. Key assumptions:

- The whole volume of drainage water is collected and recirculated, so there are no direct emissions from fertilisers to the water compartment; the leftover drainage water at the end of each crop cycle is delivered to the closest wastewater treatment plant;
- The useful lives of sand filters and UV lamps for the filtration and sterilisation units are 4 years each.

Stone wool substrate: ordinary and reuse subscenarios (sw-o and sw-r). Stone wool is used in the real-world case study. BAU combination with sw-o is the baseline for comparisons, as this is observed in the case study. In sw-o, 100% stone wool is landfilled, while packaging materials are recycled (cf. BAU). Instead, sw-r relies on the assumption that a recycling company specialised in horticultural substrates collects and recycle both the exhausted substrate and packaging (Diara et al., 2012).

Coir pith substrate: ordinary and reuse subscenarios (cp-o and cp-r). Generally, exhausted coir pith (100% mass) is spread on farmland. This is the ordinary management in Mediterranean countries which is depicted by cp-o. This is already a reuse strategy; however, the farmer may decide to deliver the exhausted coir pith (100% mass) to the closest composting plant. Packaging materials are then recycled (cf. BAU). Additional key assumptions of cp-o and cp-r:

- for the inventory analysis, detailed data for modelling the production and manufacturing of coir pith for horticulture are available for the United Kingdom (Newleaf, 2012). This study uses those data, while considering transport distances to the case study and that the final manufacturing and packaging occur in Italy (the closest plant to the case study).
- Manual labour only is required to separate the sleeves from the exhausted substrate and no additional labour force is hired.

2.2. Eco-efficiency analysis and interpretation

In the LCA, impact categories at midpoint level are selected rather than endpoint due to better consensus characterisation methods and lower statistical uncertainty (Bare et al., 2000). Impact characterisation uses ILCD 2011 Midpoint+ (EC and JRC, 2010) for climate change, ozone depletion, photochemical ozone formation, acidification, terrestrial eutrophication, marine eutrophication, freshwater eutrophication, water resource depletion, and mineral, fossil and renewable resource depletion; and USEtox 2 (recommended + interim) (Rosenbaum et al., 2008), for human toxicity cancer, human toxicity non-cancer and freshwater toxicity.

The LCA software is SimaPro 9 (Pré Consultants B.V.; licence available from the University of Pisa). In principle, LCA results depend on input data and on the impact assessment model (Gentil et al., 2010). There are two major reasons for that, i.e. the integration with and selection of supporting databases, and the implementation of impact assessment models (Lopes Silva et al., 2019). Databases are not necessarily compatible with each other, due to differences in data formatting and quality requirements, geographical and technological coverage, allocation procedures, and time relevance (Shonnard et al., 2015; Zhou et al., 2014). The implementation of impact assessment methods can result in the inclusion of different characterisation factors, with no observed consistency as to which software includes a substance and which excludes it (Speck et al., 2016).

In the LCC, the present total costs of production (TCOP) are used to evaluate economic-environmental trade-offs in the contribution to impacts of life cycle stages. As future cash flows are relevant for the assessment, which considers the greenhouse production system over its useful life and includes the end of life of all materials (Nieder-Heitmann et al., 2019). TCOP is used to calculate the net present value of discounted cash flows (NPV), using an interest rate of 10% as in similar studies (Boulard et al., 2011; Hollingsworth et al., 2020)[1]. Scenarios are economically viable when NPV>0 (scenario profitability increases with NPV). To improve the communication of findings, the EE uses the profitability index (PI), calculated as the ratio between NPV and investment costs: profitable scenarios have PI >1 and they should be preferred to the baseline when they show a greater PI of BAUsw-o.

EE uses relative values, i.e. percent change with respect to the baseline (Zhang et al., 2019). Improvement or worsening of environmental and economic indicators are plotted on a two-way graph, to identify the scenarios that are both economically and environmentally desirable (Ferrández-García et al., 2016; UNEP/SETAC, 2008). Eco-efficient scenarios show improvements in both the environmental and economic dimension, i.e. the percent change is negative for LCA impact categories (x-axis) and positive for PI (y-axis). Positive values for LCA impact categories and negative for PI pinpoint inefficient scenarios. The remaining combinations (environmental improvements, but economic worsening or the other way around) identify partially efficient scenarios.

Sensitivity analyses are carried out to estimate the effects of data choices on study findings (ISO 14040:2006). Sensitive parameters in the LCA and LCC are selected based on expert consultation and/or impact assessment findings, to support practical decision-making and limit the context-specificity of the study.

The comparison of absolute impact assessment figures with the literature largely involves LCA findings and uses studies of tomato greenhouse production in Mediterranean countries with the same system boundaries of the present research. However, the life cycle impact assessment method and the considered impact categories may differ, thereby preventing the comparison of most absolute values, but climate change (Dias et al., 2017). This is due to the large consensus among researchers on the use of the most recent characterization factors published by the Intergovernmental Panel on Climate

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[1] An interest rate of about 10% is consistent with the average internal rate of return of investments to advance agricultural systems (The Economist, 2015).

Change, with the more widespread time horizon being 100 years (Levasseur, 2015), like in the present study.

3. Data

Data collection (2019) was the most critical part of the study. The use of secondary data was limited to the background system. Multiple data sources were combined to carefully consider the similarities among the production contexts, facilities, and market conditions, and to validate data (cf. Basset-Mens et al., 2019) (Figure 2; see the Annex for a detailed description of the process).

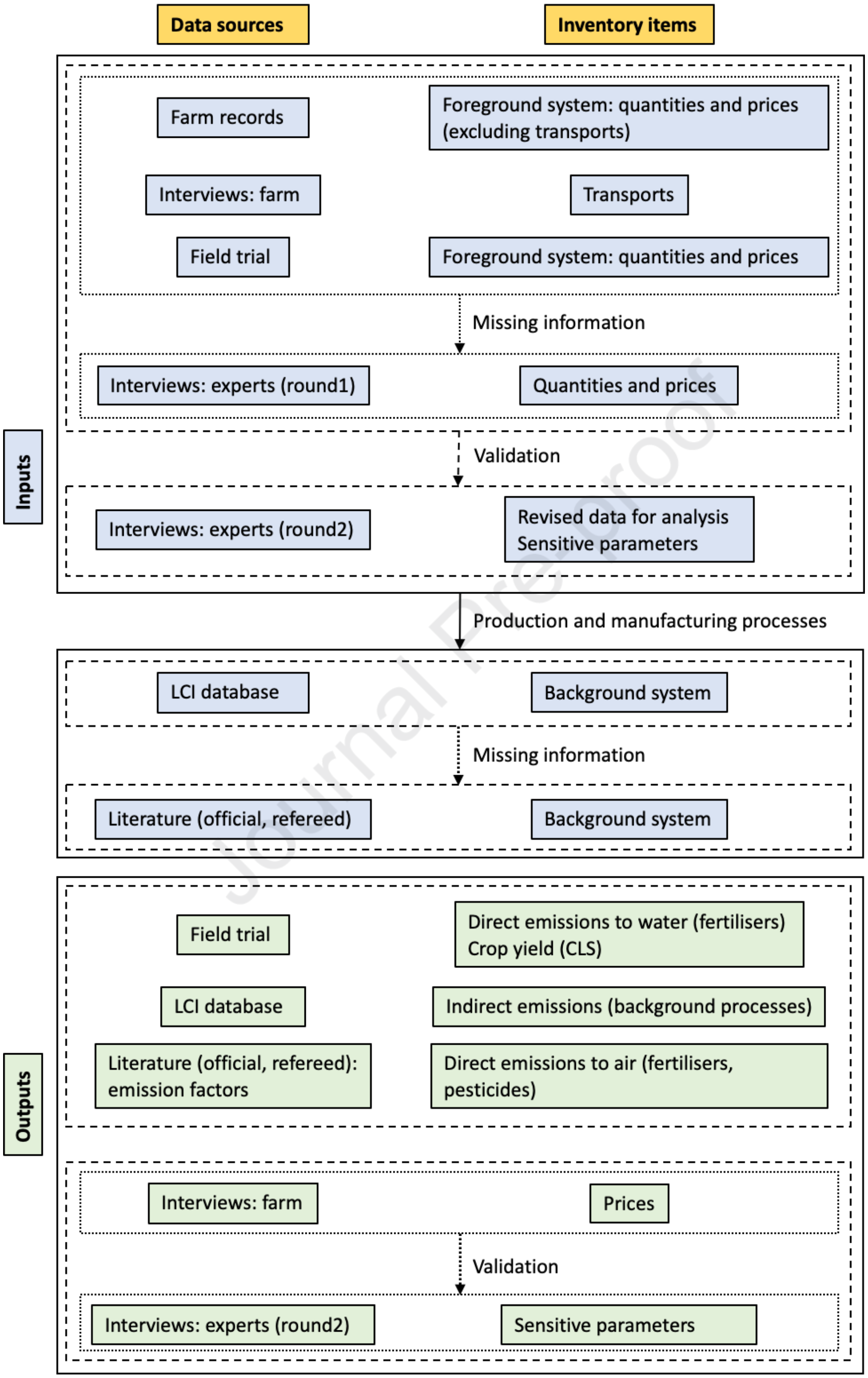

Figure 2. The data collection process. Source: Authors' own elaboration.

3.1. Case study

The case study was selected having an ongoing agreement with the University of Pisa for carrying out field experiments. The case study is located in Tuscany (central Italy), in the province of Pistoia (administrative centre 43°56′N 10°55′E)[2], an area specialised in protected agriculture (Figure 3).

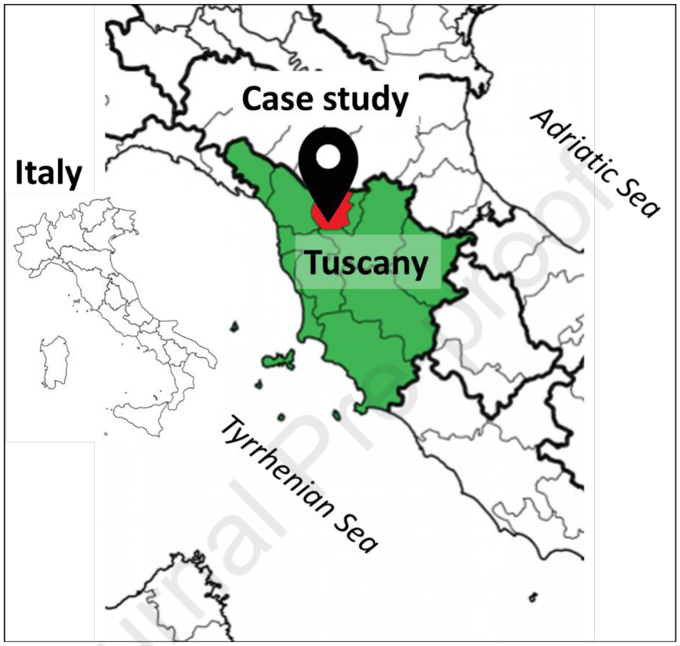

Figure 3. Case study map. Source: Authors' own elaboration.

The province has a total surface of 965km^2 and an average height above the sea level of 245m. The average annual precipitation is 1200mm, distributed over 95.1 days, and the average annual temperature is 15.3°C (Consorzio LaMMA, 2022).

According to the most recent data (CREA, 2021), the agricultural sector of Tuscany was worth over €3.2 billions in 2019, largely due to crop cultivation (61%). Among crop farms, horticultural and floricultural farms displayed the greatest gross revenues, with an average of €186,000 per farm, about 40% more than cereal and wine farms. However, horticultural and floricultural farms were the most intensive fertiliser users, with greater than average annual consumptions of nitrogen (503 kg/ha) and phosphorous (457 kg/ha) fertilisers (on average Tuscan farms used 77 kg/ha nitrogen and 47 kg/ha phosphorous fertilisers) (CREA, 2021). The agri-food sector significantly contributed to the

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[2] Italian provinces are level 3 territorial units under the Nomenclature of Territorial Units for Statistics of the EU (European Commission, 2021).

regional economy and hold a strategic role to stimulate the economic development of rural areas (IRPET, 2021), with over €2.2 billion total value added and 1.69 full-time equivalents per farm (30% more than the national average) (CREA, 2021). Most Tuscan farms were involved in local supply chains, with 54% of production inputs and 70% of outputs being, respectively supplied and demanded from within Tuscany; that demand was driven by food processors, restaurants and retailers (IRPET, 2021).

3.2. Inventory building

The total agricultural area is 2 ha divided into two locations, located 4 km apart; the distance from suppliers and waste management plants is similar between the two locations. The utilised agricultural area includes 8 multi-span tunnels, with no heating system. Greenhouse surfaces range between 500 m^2 to 2500 m^2 (length = 34-55 m; width = 16-45 m; spans = 2-5; ridge height = 4.5 m; gutter height = 2.3 m). There are two crop growing seasons per year (March-July and August-December, 264 days/year in total). Accurate fertigation to meet crop needs is guaranteed by a computerized fertigation unit and a drip irrigation system. The unit embeds light sensors and a weather station, to modulate the distribution of the solution and the opening of rooftop ventilators. Plant protection complies with Integrated Pest Management rules. Soilless tomato cultivation (on stone wool substrate) with open-loop system was introduced in 2010. Stone-wool growing bags are used for 2 harvest years in a row and disposed to landfill at end-of life. All harvested tomatoes of commercial quality (marketable yield) are sold to a local retailer, who set the price to the farmer. The residual biomass (non-marketable tomatoes and crop residues) is spread on farmland. Labour force includes two farm household members (full-time) and a full-time worker. Farm structure is in line with relevant official statistics for farms specialized in horticulture in Tuscany (European Commission, 2020e). The life cycle inventory for the LCA is built in the SimaPro software, with the support of the Ecoinvent® 3.6 database (Wernet et al., 2016) for the background system (Tables 1 through 3)[3].

Table 1. Life cycle inventory (reference period 2014-2018): material and resource inputs of all scenarios. Production and manufacturing are based on the Ecoinvent® 3.6 database (Wernet et al., 2016), when not differently stated. BAU: business as usual; WWTP: wastewater treatment plant; CSC: cascade cropping; CLS: closed-loop fertigation. EDTA: ethylenediaminetetraacetic acid. Source: Authors own elaboration.

Inputs	**Unit/ha/yr**	**Scenarios**				**Notes**
		BAU	**WWTP**	**CSC**	**CLS**	
Water	m3	8632	8632	8632	6831	
Sand	kg	-	-	-	38	

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[3] Agri-footprint® 4.0 (Blonk Consultants, 2017) and USLCI (NREL, 2012) databases were added to create missing processes in Ecoinvent® 3.6.

Inputs	Unit/ha/yr	Scenarios				Notes
Seedlings	pieces	15876	15876	15876	15876	
Cardboard	kg	46	46	46	46	Hives; packaging
Concrete	m^3	9	9	9	9	Plinth foundations, walkway; outdoor tank in WWTP, CSC = 0.0007 kg/ha/ha
Metals	kg	263	288	288	272	Steel: posts, frame reinforcements, gutters, axes, profiles, arches, ventilators, wire; outdoor tank, sand filter Cast iron: engine, pumps
Plastics	kg	1807	1956	1942	2017	LDPE: Greenhouse coverage, tomato packaging; slab sleeves; packaging; pheromone dispensers HDPE: Anti-aphid net; indoor tanks; packaging PET: Pipes, drippers, microtubes Polypropylene: Floor mulching, raffia thread; outdoor tank PVC: Plant gutter system, clips, wedges; distribution system; outdoor tank Polystyrene: Substrate layers
Stone wool	kg	1976	1976	1976	1976	Density: 46.3 kg/m^3
Coir pith (70% fibre, 30% pith)	kg	3450	3450	3450	3450	Density: 81 kg/m^3. Production and manufacturing based on (Newleaf, 2012)
Control units	kg	9.5	9.5	9.5	14.5	Fertigation/sterilisation
UV lamps	pieces	-	-	-	0.75	Sterilisation
Electricity, production mix	kWh	2018	3005	3287	5407	
Fertilisers						
Calcium nitrate	kg	617	617	617	489	
Potassium nitrate	kg	547	547	547	433	
Magnesium sulphate	kg	824	824	824	1516	
Monopotassium phosphate	kg	732	732	732	580	
Potassium sulphate	kg	1138	1138	1138	901	
Ferric EDTA	kg	167	167	167	132	Production and manufacturing based on stoichiometry
Copper EDTA	kg	2.3	2.3	2.3	1.8	Production and manufacturing excluded from the assessment (Zampori et al., 2016)
Zinc EDTA	kg	9.3	9.3	9.3	7.3	Production and manufacturing excluded from the assessment (Zampori et al., 2016)
Manganese EDTA	kg	12	12	12	9.3	Production and manufacturing excluded from the assessment (Zampori et al., 2016)
Sulfuric acid	kg	2251	2251	2251	1781	
Pesticides						
Pyraclostrobin	g	500	500	500	500	
Dimethomorph	g	900	900	900	900	
Pyrimethanil	g	800	800	800	800	
Fenhexamid	g	750	750	750	750	

Inputs	Unit/ha/yr	Scenarios				Notes
Cyprodinil	g	300	300	300	300	
Fludioxonil	g	200	200	200	200	
Methoxyfenozide	g	288	288	288	288	
Emamectin benzoate	g	43	43	43	43	
Orange essential oil	g	240	240	240	240	Production and manufacturing based on (Beccali et al., 2009)
Spirotetramat	g	192	192	192	192	
Azadirachtin	g	78	78	78	78	
Acetamiprid	g	200	200	200	200	
Bacillus thuringiensis	g	720	720	720	720	Production and manufacturing based on (Rowe and Margaritis, 2004)
Spinosad	g	240	240	240	240	
Pheromone	g	60	60	60	60	

The greenhouse (B15 class European Standard EN 13031-1:2003) has concrete foundations and walkway, a steel frame and LDPE covering. Roof and lateral windows are operated by an electric engine and manually, respectively. Tomato seedlings are sourced from a neighbouring nursery and transferred to small substrate cubes, before transplanting in the substrate (2646 slabs/ha; 3 plants/m^2).

The fertigation control unit prepares and distributes the nutrient solution via drip irrigation. The floor is covered with polypropylene mulching that, in BAU, has openings for draining the surplus nutrient solution to the ground. In WWTP, CSC, CLS a gutter system collects that surplus solution, which is pumped either to an outdoor tank (40m^3; concrete base, steel structure, plastic coverage, and interior; WWTP, CSC), or to indoor plastic tanks (6m^3 total) and subsequently through the sterilisation unit, before recirculation (CLS).

Electricity modelling is based on the Italian country mix, where the share of fossil sources is 61%, of which 13% coal (renewables = 39%; (IEA, 2018).

Due to limitations of the Ecoinvent® 3.6 database, official and refereed literature was retrieved to bridge information gaps (coir pith, orange essential oil, Bacillus thuringiensis). For missing fertiliser processes, stoichiometry was used when the process contributed for at least 5% to the impacts of the life cycle stage (ferric ethylenediaminetetraacetic acid), otherwise the relative background processes were excluded from the assessment (copper, manganese, and zinc ethylenediaminetetraacetic acid) (Zampori et al., 2016).

The materials for all greenhouse stages are sourced from farm neighbourhoods, except for pollination hives and the substrate (Table 2).

Table 2. Transport distances and means of transport. Source: Authors' own elaboration.

Materials	Distance (km/yr)	Means of transport
Greenhouse construction materials; coir pith slabs; fertilisers; pesticides	8.2	Lorry
Bumblebee hives	228	Van
Seedlings	9.5	Van
Tomato packaging	69	Lorry
Fertigation and drainage water management materials	8.4	Lorry
Stone wool slabs	334	Lorry
Construction and demolition materials to dedicated waste plant	3.2	Lorry
Waste to municipal waste sorting plant	9.9	Lorry
Drainage water to wastewater treatment plant	8.4	Tank lorry

The calculation of emissions to air from fertilisers moves from a mass balance and uses emissions factors (Nemecek and Kägi, 2007): N_2O = 1.25% and NH_3 = 2% total nitrogen applied with fertilisers; NOx = 0.21×N_2O. The calculation of emissions to air from pesticides is based on (Juraske et al., 2007) (Table 3).

Table 3. Life cycle inventory (reference period 2014-2018): outputs of all scenarios. Data are from primary sources, when not differently stated. BAU: business as usual; WWTP: wastewater treatment plant; CSC: cascade cropping; CLS: closed-loop fertigation. EDTA: ethylenediaminetetraacetic acid. Source: Authors own elaboration.

Outputs	Unit/ha/yr	Scenarios				Notes
		BAU	WWTP	CSC	CLS	
Marketable tomatoes	t	193	193	193	191	90% of gross yield. Biomass: BAU, WWTP, CSC = 92 t; CLS = 89 t; root weight included in the substrate at end-of-life
Drainage water	m^3	1682	1682	-	12.7	
Emissions to water						
NO_3	kg	251.81	-	-	-	
NH_4	kg	5.33	-	-	-	
PO_4	kg	25.4	-	-	-	
K	kg	343.5	-	-	-	
Ca	kg	303.97	-	-	-	
Mg	kg	60.49	-	-	-	
Na	kg	96.78	-	-	-	
SO_4	kg	241.69	-	-	-	
Cl	kg	71.59	-	-	-	
Fe	kg	2.12	-	-	-	
EDTA	kg	3.59	-	-	-	
Emissions to air (fertilisers)						*Calculated from input data based on*
N_2O	kg	3.33	3.33	3.33	2.09	
NH_3	kg	5.33	5.33	5.33	3.35	
NOx	kg	0.7	0.7	0.7	0.44	
Emissions to air (pesticides)						*Calculated from input data based on (Juraske et al., 2007)*
Pyraclostrobin	g	2.5	2.5	2.5	2.5	

Outputs	**Unit/ha/yr**	**Scenarios**				**Notes**
Dimethomorph	g	4.5	4.5	4.5	4.5	
Pyrimethanil	g	4	4	4	4	
Fenhexamid	g	3.75	3.75	3.75	3.75	
Cyprodinil	g	1.5	1.5	1.5	1.5	
Fludioxonil	g	1	1	1	1	
Methoxyfenozide	g	1.44	1.44	1.44	1.44	
Emamectin benzoate	g	0.21	0.21	0.21	0.21	
Orange essential oil	g	1.2	1.2	1.2	1.2	Characterization factor added to the USEtox model from (OLCA-Pest project, 2021)
Spirotetramat	g	0.0096	0.0096	0.0096	0.0096	
Azadirachtin	g	0.39	0.39	0.39	0.39	
Acetamiprid	g	1	1	1	1	
Bacillus thuringiensis	g	0.036	0.036	0.036	0.036	
Spinosad	g	1.2	1.2	1.2	1.2	
Pheromone	g	0.003	0.003	0.003	0.003	

The economic inventory (LCC) is built using a Microsoft Excel® spreadsheet. Data cover the total production costs over the greenhouse life cycle (purchase, use and end of life management) and revenues (BAU, WWTP, CSC = € 208494 /ha/yr, CLS = € 206388 /ha/yr) (Table 4).

Table 4. Life cycle costs and revenues. Source: Authors' own elaboration.

Costs (€/ha/yr)	**BAU**	**Other scenarios/subscenarios (different figures only)**
Greenhouse		
Investment (including project design)	12600	WWTP, CSC = 12925; CLS = 12800
Maintenance	1080	WWTP, CSC, CLS = 1830
Consumables	57311	CLS = 56921
Transport (consumables only)	5.6	
Electricity	5.4	
Advisory and administration	1000	
Labour	15500	
Fertigation		
Investment	1000	WWTP, CSC = 1500; CLS = 3200
Maintenance	3300	
Drainage water management	0	WWTP = 9890; CLS = 75
Electricity	528	WWTP = 789; CSC = 863; CLS = 1423
Substrate		
Stone wool	3969	
Coir pith	5821	
Fertilisers		
Consumables	5121	CLS = 4053
Transport	18.6	

Costs (€/ha/yr)	BAU	Other scenarios/subscenarios (different figures only)
Pesticides		
Consumables	421	
Transport	0.2	
Waste		
Greenhouse demolition	70.1	
Plastics sw-o, cp-o, cp-c	186	WWTP, CSC = 193; CLS =194
Plastics sw-r	153	WWTP, CSC = 159; CLS =160
Substrate sw-o	22117	sw-r, cp-o = 0; cp-c = 6909
Fertigation/sterilisation units	1.3	CLS = 2.1
Other waste	33	CLS = 37.0

Farmer prices are already charged with the prices for background processes, e.g. delivery, construction, assembly, to cite a few (Heijungs et al., 2013). Investment costs involve the materials for building the greenhouse and fertigation infrastructures. Project design, construction fees, overhead costs farm advisory, and labour are allocated to the greenhouse infrastructure stage. Variable costs include utilities, consumables and waste and the relative transports. Farmer price for marketable tomato is €1.2/kg, subject to 10% value added tax (Italian consumption tax system) (cf. Testa et al., 2014b).

In the environmental and economic inventories (Tables 1 through 4), key differences of reuse scenarios are in the fertigation and fertilisers stages, as follows:

- WWTP, CSC: greater quantities of construction materials for building the outdoor tank;
- WWTP: higher costs due to the fees for wastewater management;
- CLS: greater quantities of electronic components and plastics for building and operating the closed-loop system;
- CLS: over 20% water and fertiliser savings.

Parameters for sensitivity analyses

Based on impact assessment results, a sensitivity analysis is carried out to evaluate the extent to which extending the lifespan of the greenhouse and fertigation infrastructures from 20 years to 25 and 30 years would affect environmental impacts (Bartzas et al., 2015; Boulard et al., 2011).

Transport distances are sensitive parameters identified via expert interviews, as those observed in the case study are shorter than in most farms; increasing those distances by 50% and 100% would improve the understanding of the extent to which transport distance contribute to environmental impacts.

The impact of electricity is identified as an important environmental aspect via expert interviews, as greenhouses have been more reliant on electronic components through time. To consider that, a sensitivity analysis is carried out on electricity production. The shares of renewable and fossil resources are varied to consider 2030 and 2050 targets of Italy's National Energy Strategy (i.e. phasing out of coal by 2030 and progressive reduction of fossil sources to 40% in 2030 and 7% in 2050; MATTM, 2017).

Price adjustment up to ±20% by retailer companies (key buyers) through time is identified as a key economic problem for decision makers. A sensitivity analysis is carried out to evaluate the extent to which price fluctuations of ±5%, ±10%, ±20% affect the economic viability of each scenario-subscenario combination.

4. Results

4.1. Life cycle assessment and life cycle costing

Study findings show that upgrading BAU to collect and reuse drainage water for agricultural purposes on farm (CSC, CLS) or indirect uses off farm (WWTP) abates marine and freshwater eutrophication (Table 5).

Table 5. Assessment results per scenario per functional unit (1 ha greenhouse): characterized life cycle impacts per year, total costs of production (TCOP) per year, net present value (NPV) and profitability index (PI) over the lifetime of the greenhouse (20 years). Source: Authors' own elaboration.

	BAU	**WWTP**	**CSC**	**CLS**
Life cycle assessment				
CC (kg CO_2 eq, 100 years)	1.93E+04	2.291E+04	2.08E+04	2.09E+04
OD (kg CFC-11 eq)	1.48E-03	1.85E-03	1.58E-03	1.58E-03
PM (kg PM2.5 eq)	1.25E+01	1.52E+01	1.40E+01	1.40E+01
POF (kg NMVOC eq)	5.92E+01	7.15E+01	6.32E+01	6.28E+01
AC (molc H+ eq)	1.55E+02	1.79E+02	1.66E+02	1.62E+02
TE (molc N eq)	3.18E+02	3.84E+02	3.45E+02	3.54E+02
FE (kg P eq)	1.38E+01	9.05E+00	7.17E+00	7.40E+00
ME (kg N eq)	6.86E+01	5.73E+01	2.15E+01	2.17E+01
LU (kg C deficit)	3.23E+04	4.25E+04	3.41E+04	3.35E+04
WRD (m3 water eq)	6.09E+03	7.50E+03	7.56E+03	6.00E+03
MFR (kg Sb eq)	5.09E+00	5.51E+00	5.42E+00	5.23E+00
HTC (cases)	1.35E-03	1.81E-03	1.48E-03	1.56E-03
HTnC (cases)	5.46E-03	9.25E-03	5.32E-03	6.02E-03
FET (PAF.m3.day)	8.76E+07	1.13E+08	8.48E+07	9.73E+07
Life cycle costing				
TCOP (€)	1.26E+05	1.39E+05	1.29E+05	1.33E+05

	BAU	WWTP	CSC	CLS
NPV 20 years (€)	5.28E+05	3.93E+05	4.76E+05	4.38E+05
PI 20 years	2.02E+00	1.44E+00	1.74E+00	1.52E+00

Compared to BAU, freshwater eutrophication (FE) and marine eutrophication (ME) are, respectively, -34% and -16% when the drainage water is treated in a wastewater treatment plant (WWTP), -48% and -69% when drainage water is recycled on a second crop (CSC), and -46% and -68% when drainage water is recirculated on the same crop (CLS) (Parada et al., 2021).

Despite water savings, water resource depletion (WRD) does not decrease markedly (-1%) in CLS compared to BAU, as in BAU the entire volume of fertigation water is released to the environment.

In BAU, TCOP confirms previous research (Llorach-Massana et al., 2016). Reuse scenarios increase TCOP between 2% (CSC) and 10% (WWTP) and reduce PI compared to BAU, especially WWTP (-29%) and CLS (-25%), though keeping their profitability (PI>1). The reduction of PI in reuse scenarios contrasts with the findings of similar studies (Galdeano-Gómez et al., 2017).

Different to ME and FE, terrestrial eutrophication (TE) increases, especially in WWTP (+21%), as this impact depends more on emissions to air (Posch et al., 2008; Seppälä et al., 2006). Compared to BAU, direct emissions to air differ slightly in CLS only, while indirect emissions increase in all reuse scenarios due to the greater material quantities.

Acidification (AC) is directly related to the applied quantity of fertilisers (Muñoz et al., 2008) and, like TE, depends on emissions to air (especially NH_3, NO_2, SO_x) (Posch et al., 2008; Seppälä et al., 2006). Reuse scenarios do not allow the reduction of TE and AC.

The impact on climate change is close to (Martínez-Blanco et al., 2009) or lower than (Payen et al., 2015; Torrellas et al., 2012) similar studies. Reuse scenarios increase the remaining impact categories (PM, OD, POF, LU, MFR, HTC, HtnC, FET), compared to BAU. Environmental worsening is moderate in CLS, ranging between 3% (MRF) and 16% (HTC), but it is more relevant in WWTP, especially for land use and toxicity (+32% LU, +34% HTC, +69% HTnC, +29% FET), due to the large volume of chemically-treated drainage water in the wastewater treatment plant (Linderholm et al., 2012). Just CSC can reduce toxicity impacts (ca. -3% HTnC and FET).

The contribution analysis emphasises the effect of the fertigation and fertilisers stages on impact assessment results (Figure 4).

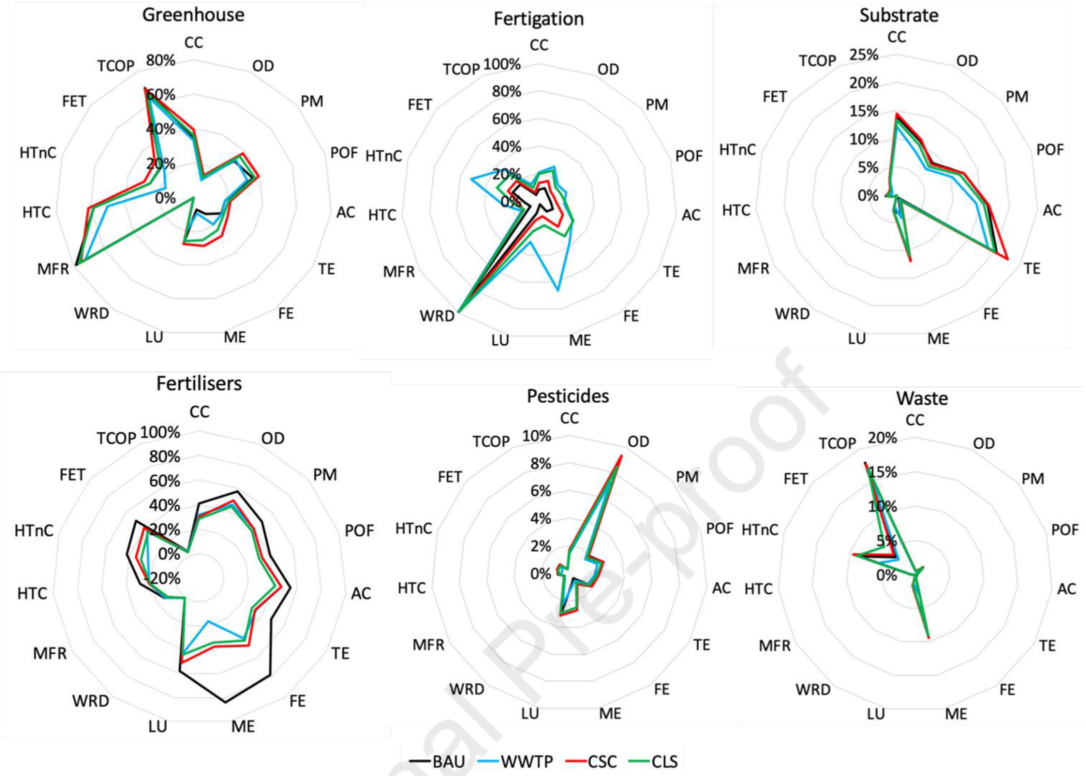

Figure 4. Life cycle stage contribution to environmental impacts and TCOP. Source: Authors' own elaboration.

The effect of the wastewater treatment plant is pinpointed by the contribution of the fertigation stage to toxicity impacts in WWTP (HTnC = 52%, FET = 35%, HTC = 26%) and CLS (HTnC = 32%, FET = 35%, HTC = 18%), compared to BAU. In WWTP, toxicity impacts are caused by industrial processes to produce plastics and construction materials (HTC) and the wastewater treatment plant (HTnC, FET).

In reuse scenarios, the fertigation stage increases TCOP as well, especially in WWTP (1.8 times BAU) and CLS (1.2 times BAU).

Compared to BAU, the fertilisers stage contributes -59% to ME (7.4 kg N eq) in CLS and -81% (9 kg N eq) in WWTP; while ME of the fertigation stage is 2.5 times BAU in CLS (3.9 kg N eq) and 10 times CLS in WWTP (39 kg N eq). Similar reasoning applies to FE; the absolute FE values for the fertigation and fertilisers stages are as follows: fertigation, BAU = 1.3 kg P eq, WWTP = 3.4 kg P eq, CSC = 1.5 kg P eq, CLS = 2.3 kg P eq; fertilisers, BAU = 11 kg P eq, WWTP, CSC = 3.7 kg P eq, CLS = 3.2 kg P eq.

Study findings confirm previous research (Martínez-Blanco et al., 2011; Testa et al., 2014a), by identifying the greenhouse stage as the major source of environmental and economic impacts in all scenarios, especially with respect to TCOP (€ 87502-88252) mainly due to consumables, CC (6660-7316 kg CO_2 eq), MFR (4 kg Sb eq), HTC ($9 \cdot 10^{-4}$ cases) mainly due to the production of construction materials and electricity (CC).

Other life cycle stages are minor, with reuse scenarios not deviating much from BAU, as in previous research (Torrellas et al., 2012). Direct emissions from pesticides contribute substantially to toxicity impacts (Schmidt Rivera et al., 2017). A possible explanation for the reduced contribution of pesticides is the adoption of Integrated Pest Management. Most environmental impacts of substrate (stone wool) are generated during manufacturing and emissions after landfilling (cf. Savvas and Gruda, 2018). TCOP (€ 3969 substrate; € 22408-22420 waste) depends to a great extent on the purchase and landfilling of stone wool.

Compared to landfilling, recycling exhausted stone wool allows slight environmental improvements, but great cost savings (-18% TCOP) (Figure 5).

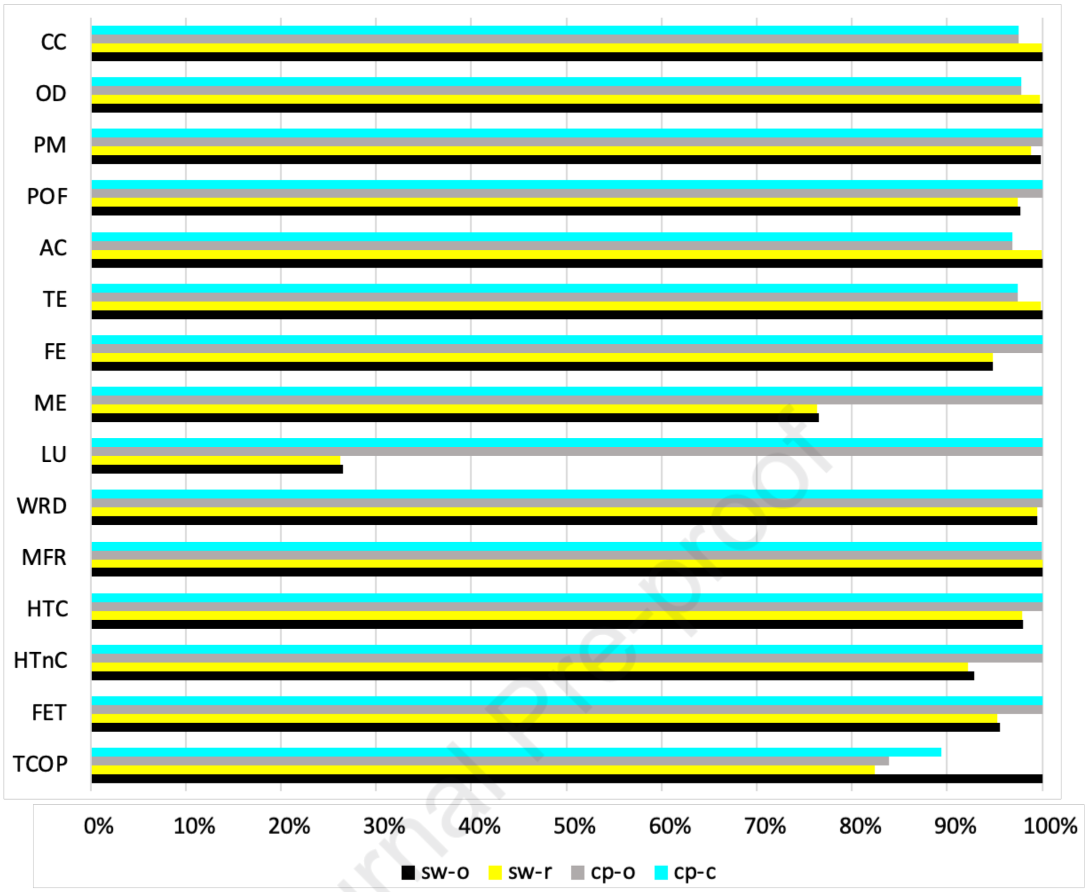

Figure 5. Contribution to LCA impact categories and LCC of different subscenarios. Source: Authors' own elaboration.

Coir pith is biodegradable on farm (land spreading) or via composting; so, cp-o and cp-r subscenarios do not include landfilling. Shifting to coir pith allows small environmental improvements in some impact categories (2% to -3% of CC, OD, AC, TE) and relevant environmental worsening in other impact categories, as shown elsewhere (Antón et al., 2005a; QUANTIS, 2012). Especially, ME and LU grow by ca. +30% and +300%, respectively, due to land occupation by coconut plantations and emissions from fibre and pith processing. Despite the higher purchase cost (€ 2.2/coir pith slab vs. € 1.5/stone wool slab), shifting to coir pith decreases TCOP up to -16% (sw-o), by abating the disposal fees at end of life (€ 1.4/kg for landfilling stone wool, € 0.8/kg for composting coir pith, no cost for spreading coir pith on land and recycling stone wool). Subscenarios alternative to the baseline (sw-o) have a negligible effect on PM, MFR, and WRD.

Sensitivity analyses

The extension of the useful life of the production facilities up to 30 years reduces all absolute impact figures, with no remarkable differences among scenarios (Figure 6).

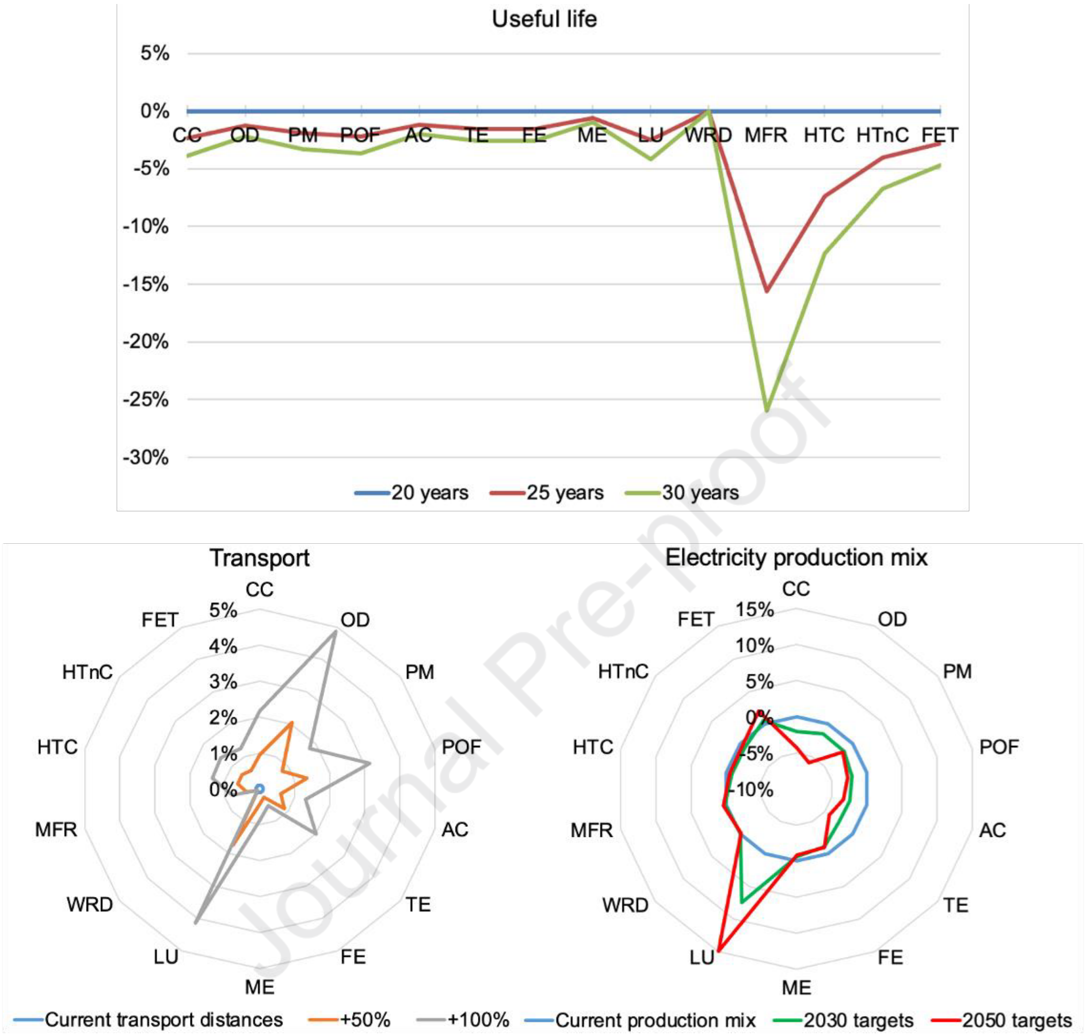

Figure 6. Sensitivity of characterised impacts (BAU) to increased lifespan and transport distance, and future changes in the Italian electricity mix. Source: Authors' own elaboration,

CC and OD decrease, as in similar studies (Bartzas et al., 2015), though the greatest environmental impact mitigation potential is in terms of MFR (> 25% reduction) and HTC (> 12% reduction), due to the smaller quantities of construction materials.

Transportation does not contribute much to environmental impacts (cf. Bartzas et al., 2015), as this study is limited to the farm gate (Page et al., 2012), then even doubling distance has not major effects on the overall impact of transports (max increase: OD = about +5%).

Future changes in the production of the Italian electricity mix can have a marked effect on LU (up to +15% in 2030), due to the increase of photovoltaic mounting systems. Other environmental impacts are expected to decrease up to -5% (OD).

Changes in producer price for tomatoes markedly affect the economic viability of scenario-sub-scenario combinations (Figure 7).

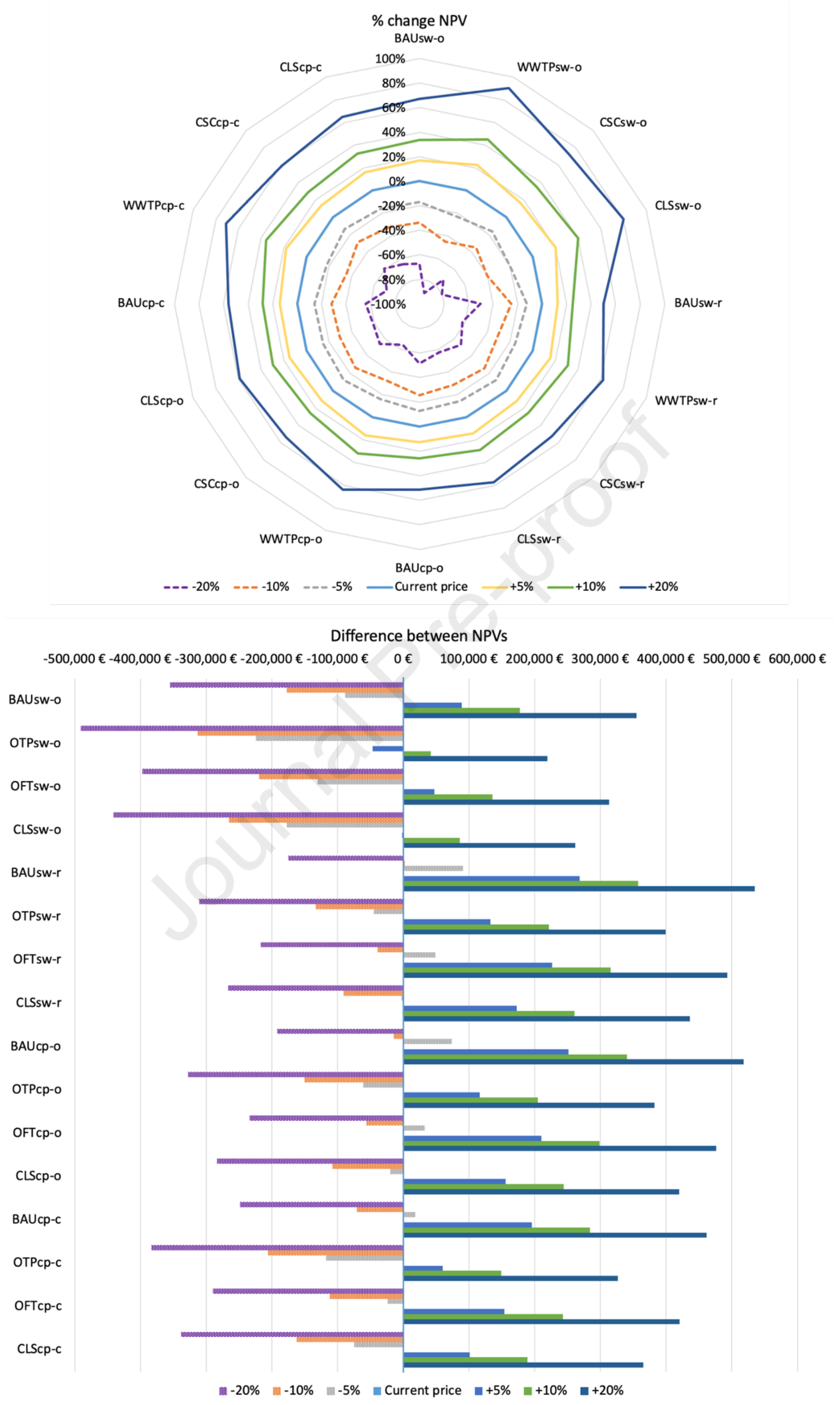

Figure 7. Changes in producer price: sensitivity of net present value (NPV) and scenario viability with respect to the baseline (BAUsw-o). Source: Authors' own elaboration.

When considered per se, all scenarios but WWTPsw-o, would be viable even with -20% producer price. Instead, no scenario would be viable compared to the baseline with the same reduction of the producer price. WWTPsw-o, CLSsw-o display the largest NPV fluctuations and would require at least +10% producer price to be viable, while a 5% increase would be enough for the rest of scenarios. CSCsw-r, CSCcp-o, CLScp-o, and BAUcp-r would be viable even with -5% producer price and BAUsw-r even with -10%.

4.2. Eco-efficiency analysis

EE emphasises the trade-offs among impact categories (Figure 8).

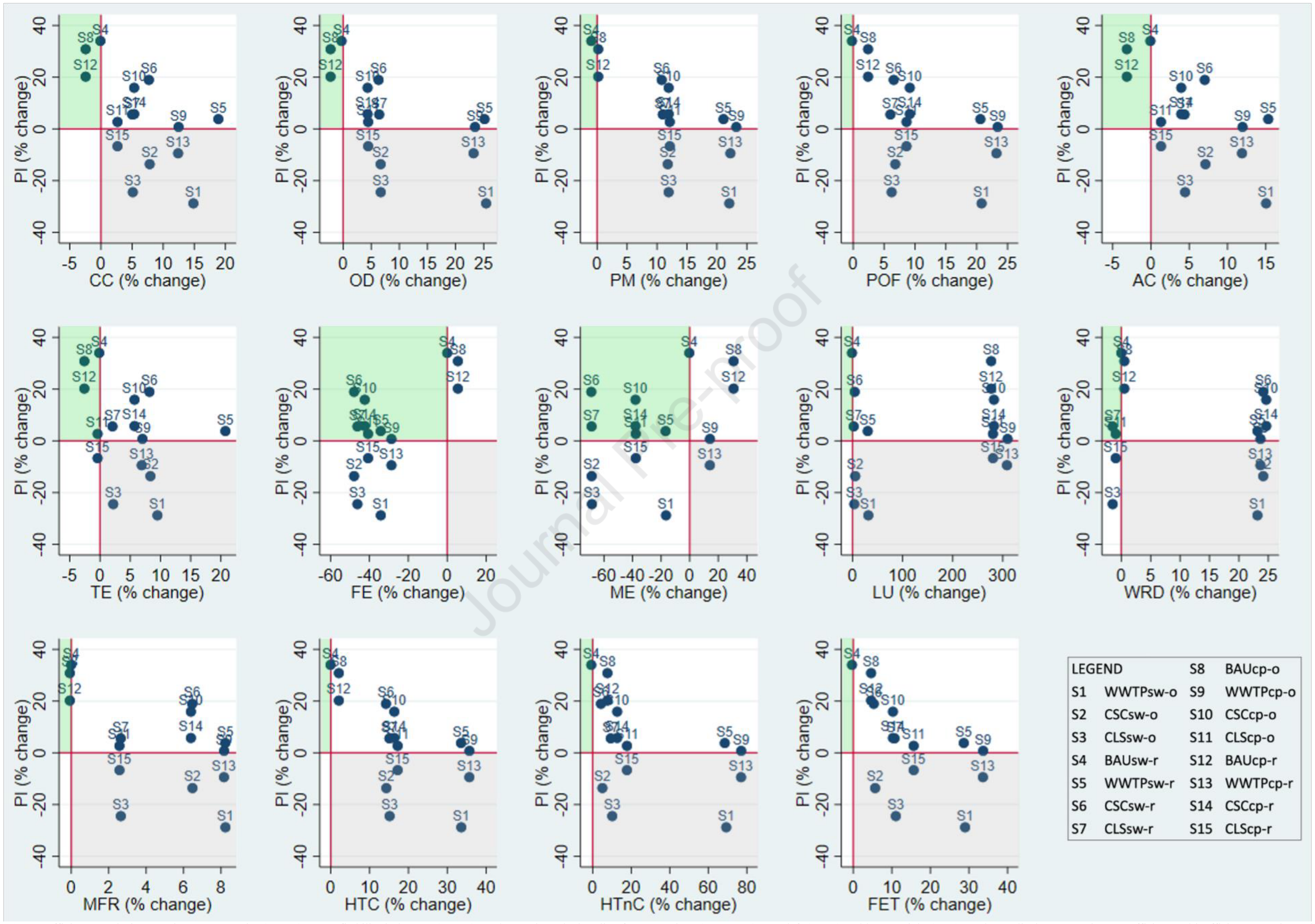

Figure 8. Eco-efficiency analysis: LCA impact categories (x-axes) vs. profitability index, PI, (y-axes). Red axes show the baseline for comparison (BAUsw-o). Top left (green) and bottom right (grey) quadrants display, respective eco-efficient and inefficient scenarios; other quadrants show partially efficient scenarios. Source: Authors own elaboration.

The x-axis identifies the scenarios that are economically acceptable for the farmer (PI change > 0%) and those that are not. Study findings suggest that adopting a reuse strategy would make sense just if coupled with changes to the substrate material (here coir pith instead of stone wool) and/or in the way how exhausted slabs are managed at end of life. Then, WWTPsw-o, CSCsw-o, CLSsw-o are generally inefficient. The changes simulated by subscenarios have the potential to reduce TCOP, by avoiding landfilling fees throughout the greenhouse lifetime, thereby increasing PI up to 34% (BAU-sw-r). This explains why BAU-subscenario combinations alternative to the base line (BAUsw-o) are eco-efficient with respect to a series of impact categories, but ME, FE, and WRD. However, the potential environmental improvements are very low, especially in BAUsw-r (almost neutral), and never exceed -3% (AC in BAUcp-o, BAUcp-r) reduction in the absolute values of characterised impacts. Other generally inefficient scenarios are WWTPcp-r and CLS-cpr. TCOP increase in WWTP and CLS compared to BAU due to the adoption of the reuse technology makes the abatement of substrate-related costs (sw-r, cp-o) necessary to raise PI and then encourage farmer uptake.

As expected from LCA results, water reuse technologies are eco-efficient with respect to FE and ME, when coupled with the adoption of reuse strategies for substrate management. WWTPcp-o and WWTPcp-r are exceptions, being eco-efficient just in terms of FE. The contribution of eco-efficient scenario-subscenario combinations to reduce FE and ME while increasing PI, compared to the baseline, are as follows: WWTPsw-r (FE = -34%, ME = -17%, PI = +3.8%), CSCsw-r (ME = -48%, FE = -69%, PI = +19%), CLSsw-r (FE = -46%, ME = -69%, PI = +5.6%), CSCcp-o (FE = -42%, ME = -38%, PI = +16%), CLScp-o (FE = -41%, ME = -38%, PI = +2.7%), CSCcp-r (FE = -42%, ME = -38%, PI = +5.8%).

Study findings highlight that reuse scenarios do not allow to achieve eco-efficiency in terms of other environmental impact categories. The only exception to this pattern is CLS in case stone wool is replaced with coir pith, which in turn is spread on farmland at the end of its useful life. If the farmer decides to keep the business as usual, eco-efficiency can be achieved in terms of CC, OD, PM, AC, TE, MFR.

Concerning WRD, adopting CLS is the only way to enable eco-efficiency, though with relatively little environmental improvement. Again, subscenario matters, by affecting both environmental and economic impact indicators. CLSsw-r and CLScp-o can, respectively, reduce WRD with -1.5% and -1%, and increase PI with 5.5% increase of PI with 2.5%; CLSsw-o and CLScp-r are just partially eco-efficient due to the high fees for stone wool landfilling and coir pith composting at end of life, respectively.

CSC and CLS are eco-efficient reuse strategies to reduce the critical impacts of greenhouse cropping (FE, ME) (cf. Martin-Gorriz et al., 2020; Rufí-Salís et al., 2020b). While CLS only allows

WRD reduction (about -1%), CSC offers greatest returns to the farmer (PI increases up to 19%). The eco-efficiency of both scenarios, however, occurs when the substrate is not landfilled and there is no disposal fee for the farmer. Profitable substrate management alternatives for the farmer involve (i) using stone wool and delivering the exhausted substate to a recycling plant, or (ii) using coir pith and spreading the exhausted substrate on farmland.

5. Discussion

5.1. Key research findings

Study findings show that reusing drainage water can mitigate FE and ME of soilless greenhouse cropping, especially due to the reduction of the contribution of the fertiliser stage to environmental impacts, coherently with the literature (Rufí-Salís et al., 2020b). Largely, this is because reuse strategies prevent nitrogen and phosphorus emissions to water (Antón et al., 2005a). TE does not deviate much across the evaluated scenarios, being associated with direct emissions to air.

When the adoption of closed-loop fertigation is considered within the whole farm economy, reduced eutrophication impacts can add to other environmental improvements due to a more efficient use of water and fertilisers (Clark and Tilman, 2017; Montero et al., 2009). However, the reduction of ME and FE comes at greater economic costs (TCOP) compared to the baseline (BAU scenario). Reuse scenarios reduce PI, due to the high investment costs for the technological upgrade, maintenance costs (electronic components should be replaced every 10 years) and wastewater treatment fees, which are not compensated by greater returns from product sale. This finding contrasts with other literature, which suggests the existence of reinforcing feedback loops between the optimisation of fertiliser and water inputs and the improvement of the economic performance of Mediterranean greenhouses (Galdeano-Gómez et al., 2017).

Reusing drainage water through closed-loop fertigation does not deliver a marked reduction of WRD compared to BAU. This is probably due to the selected life cycle impact assessment method. In the ILCD method, WRD is estimated via a scarcity model (Frischknecht et al., 2009), which considers the volume of water withdrawal and replenishment in an area and provides and indicator for the deprivation of water resources to users in that area (Boulay et al., 2015).

In general, the relatively low CC found here might be due to the different assumptions about the lifespan of the greenhouse and fertigation infrastructures, thereby pointing to the relevance of proper maintenance for extending the lifetime of infrastructures (Parajuli et al., 2019).

EE is a useful tool to identify the relevant technological options for consideration by decision makers and policy makers. Per each environmental impact category, eco-efficient scenarios can simultaneously reduce environmental impacts and increase of the profitability of the investment, compared to the status quo (Figure 9).

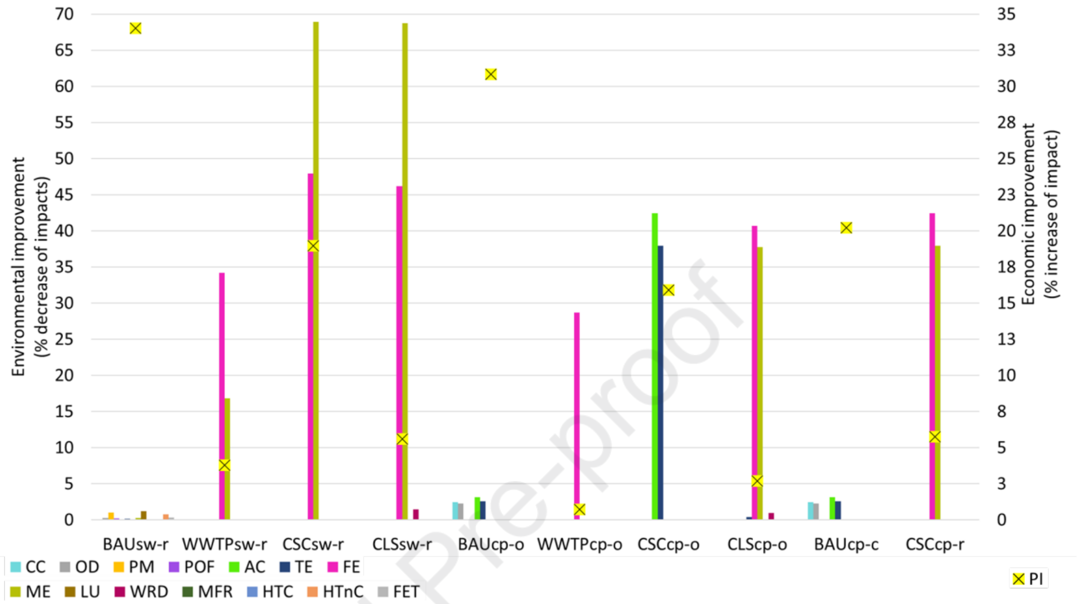

Figure 9. Eco-efficient scenarios. Environmental improvement is calculated as the negative of % change per each impact category. LCA impact categories are displayed on the primary y-axes (left-hand side); PI is displayed on the secondary y-axes (right-hand side). Source: Authors' own elaboration.

Results suggest that small changes to the status quo can markedly increase farm profit, i.e. by just modifying substrate management at end of life (BAUsw-r) or by replacing inorganic with organic substrate, as well (BAUcp-o, BAUcp-r). To enable marked reductions of acidification and, especially, eutrophication deeper changes are needed in the greenhouse technology, i.e. water reuse strategies should be adopted. Both CSC and CLS are eco-efficient alternatives to the current production technology, provided that the substrate is not landfilled and there is no disposal fee for the farmer. For example, exhausted stone wool slabs can be delivered to a recycling plant, while exhausted coir pith can be spread on farmland. Delivering drainage water to a wastewater treatment plant is another option, which could be relevant for consideration by decision-makers, for example when contextual conditions (e.g. poor farmer knowledge about other technologies) prevent the proper management of CSC or CLS systems.

The findings of this study show that increasing the efficiency of use of fertilisers and water through circular processes can have positive environmental and economic implications for the greenhouse sector, which is especially important to guarantee continuous production against sudden shortage of inputs or growth of farmer costs. With that respect, supporting research and innovation to foster technological change in greenhouses is required to trigger farmer behaviour towards the sustainable transformation of intensive food production systems (Sarabia et al., 2021).

The development and findings of the presented research suggest a series of lessons learnt with theoretical and policy implications beyond the case study level (Yin, 2014).

5.2. Theoretical implications

The contribution of this article to the literature is twofold:

1. Methodologically, the article sustains the use of LCC within EE, by basing the analysis on co-developed and methodologically consistent LCA and LCC and aims at helping method harmonisation; especially: (i) by comparing improvement scenarios with the baseline, using relative values of all the LCA impact categories under study and of the economic indicator; (ii) by using the profitability index (calculated based on discounted cash flows and investment costs) as the economic indicator; (ii) by supporting the graphical representation of EE to enable the straightforward understanding of study findings, plotting all the scenarios under evaluation on a two-way graph.
2. Content wise, the article (i) is an important data source for further research, by providing a detailed inventory of production inputs and outputs for all the evaluated alternatives; (ii) bridges a gap in the literature by showing comprehensive evidence about the environmental-economic implications of keeping the conventional open-loop fertigation technology vs. adopting three alternative reuse strategies for drainage water (treatment in a wastewater treatment plant, cascade cropping, closed-loop fertigation) and the substrate; (iii) compares inorganic vs. organic substrates and develops what if situations to show the extent to which reusing the substrate can improve the eco-efficiency of the greenhouse; (iv) supports policy design and decision making to encourage the uptake of reuse strategies in the short or mid-term by focusing on incremental technologies that are readily available on the market.

This contribution has been achieved via a challenging data collection process, to achieve the required data granularity for modelling the differences between the compared technologies. Data collection and validation relied on an accurate protocol developed by the research team and on the establishment of trusted relationships with farmers, advisors and supply chain actors (Hellweg and Mila i Canals, 2014).

5.3. Policy implications

Achieving UN Sustainable Development Goals requires the coordination of food chain actors, towards shared broad objectives. A strand of literature has highlighted the need for a radical change in the way how food is produced (Ruben et al., 2021). However, incremental innovation can offer great opportunities to improve the sustainability of food production, as well, as shown in this article. Eco-efficient reuse strategies for greenhouse production can improve the environmental and economic performance of fresh vegetable production (UNEP, 2017; Zhou et al., 2021). The findings of this research suggest that effective strategies could rely on the promotion of incremental innovation to foster the reuse of drainage water and cultivation substrates. In the European Union, the European Innovation Partnership 'Agricultural Productivity and Sustainability' set up a focus group of experts, to raise awareness about the opportunities of the diffusion of reuse strategies in Mediterranean greenhouse (EIP-AGRI, 2019a, 2019d). However, reuse technologies are not widespread in Mediterranean countries (Incrocci et al., 2020), despite their availability on the market (Massa et al., 2020, 2010). Two critical barriers have prevented their diffusion, i.e. the high uptake costs of technology and the lack of an effective knowledge network for mitigating farmers' risk aversion (EIP-AGRI, 2019b; Juntti and Downward, 2017).

5.3.1. Recommendations to address the cost barrier

To remove the cost barrier, this article compares the environmental-economic trade-offs of incremental technologies, the adoption of which can be modulated based on context-specific factors, including the ease of access to loans for the farmer (Norman and Verganti, 2014; Pearce et al., 2018). This findings aim at reducing uncertainty in policy making to encourage the adoption of reuse strategies in Mediterranean greenhouses (Herrero et al., 2021). The observed ability of the alternatives to open-loop fertigation to reduce freshwater and marine eutrophication sustains the endorsement by the European Innovation Partnership for Agricultural Productivity and Sustainability (EIP-AGRI) of reuse technologies as strategies to reduce the environmental burden of greenhouse vegetable production in Mediterranean countries, while not affecting farming viability (EIP-AGRI, 2019a, 2019e). The findings of this study present cascade cropping as a promising alternative to the status quo, by offering the greatest opportunities for improving the environmental-economic impacts of greenhouses. There is a need for the commitment of policy makers and extension services to implement adequate supporting instruments, educational campaigns and training to support the diffusion cascade cropping. Considering the water emergency, study findings point to the shift to closed-loop fertigation as a strategy for reducing the burden of greenhouse cropping on water resources. Then this technology should be considered by policy makers for improving the delivery of more sustainable fresh vegetables in water scarce areas and where climate change is projected to significantly affect the water balance (Rocha et al., 2020). However, the evidence presented in this article pinpoints trade-offs among environmental and economic impacts, similar to recent research

(Martin-Gorriz et al., 2020; Rufí-Salís et al., 2020b). Especially, (i) technological innovation to reduce the eutrophication potential can increase the climate change potential, which should be a matter of concern for policy makers, given the growing climate emergency (IPCC, 2021); and (ii) findings about the economic profitability of reuse strategies contrast with other literature, which suggests the existence of reinforcing feedback loops between the optimisation of fertiliser and water inputs and the improvement of the economic performance of Mediterranean greenhouses (Galdeano-Gómez et al., 2017). More research is still needed that integrates economic and environmental assessments into ready to use decision tools for decision makers and policy makers (Gava et al., 2020). In the EU, this is of utmost importance in the framework of the European Union’s Circular Economy Action Plan and the Farm to Fork Strategy. Reusing exhausted coir pith on farmland is already feasible and is a common practice in the case study area and in similar contexts. Instead recycling stone wool requires dedicated plants. Key producers of stone wool substrates have implemented producer take-back programmes (see e.g. Grodan (ROCKWOOL B.V.), 2017). To the best of authors’ knowledge, no similar programme has been activated in the case study area, so far. This suggests that targeted extended product responsibility legislation for the horticultural sector might have a high potential to boost the diffusion of eco-efficient technology (Galati et al., 2020). More specific agricultural policy instruments can sustain the diffusion of eco-efficient reuse technologies in Mediterranean greenhouses, by remunerating farmers based on the achieved environmental improvements, as e.g. results-based payments of the coming Common Agricultural Policy post-2020. In the European Union, new policy tools might mitigate north-south differences in the diffusion of reuse technologies as well (Thompson et al., 2020), as e.g. the recently enforced Water Reuse Regulation and the coming Integrated Nutrient Management Action Plan co-developed with Member States (European Parliament and Council, 2020).

5.3.2. Recommendations to address the knowledge network barrier

Coping for the lack of an effective knowledge network requires the improvement of the local Agricultural Knowledge and Innovation System. Even though understanding how to bridge this gap is beyond the scope of the presented research, this article could be a starting point for further studies based on stakeholder involvement in participatory activities. This could improve the existing evidence by generating a science-policy society dialogue about the multiple aspects of the production context (e.g. spatial variability, local conditions, decision makers' opinions) that are required for effective policy making. Involving stakeholders in participatory activities could have the double benefit of improving the social capital and generating input for planning new policy action. For example, participatory activities may support the understanding of the implications of technological change in socio-ecological systems. This would shade light on the interactions between biophysical elements and governance mechanisms associated with the adoption of reuse strategies in greenhouse farming (Le Moal et al., 2019). Other participatory activities may involve multi-criteria analysis workshops, where stakeholders are asked to express their opinion about multiple assessment indicators and trade-offs. This enables a greater contextualisation of findings that can provide useful inputs to design of innovative incentive mechanisms for farmers (De Luca et al., 2017).

6. Conclusions

This study shows how reuse strategies can improve the environmental and economic performance of greenhouse farming, by providing evidence from a multiple scenario analysis based on real-world data from a Mediterranean case study. A comparative eco-efficiency analysis is carried out over sixteen scenarios, by adopting a life cycle perspective. The scenarios represent different combinations of what if situations with respect to drainage water and substrate management. The purpose is to shed light on the potential sustainability improvement (or worsening) achievable through the adoption of incremental reuse technologies.

Eco-efficient technological innovation in greenhouse cropping requires the application of reuse strategies to the management of both drainage water and the growing substrate. Replacing open-loop fertigation with cascade cropping or close-loop fertigation enables almost 20% increase of the profitability index, while simultaneously reducing freshwater eutrophication (from 1.38E+01 kg P eq to 7.17E+00 kg P eq or 7.40E+00 kg P eq, respectively) and marine eutrophication (from 6.86E+01kg P eq to 2.15E+01 kg P eq and to 2.17E+01 kg P eq, respectively), when the exhausted substrate is reused (complete recycling of stone wool) at end of life and there is no disposal fee for the farmer (using coir pith as soil amendment).

There are important trade-offs among impact categories, with the compared technologies having the potential to increase a series of environmental impact categories, such as, e.g., toxicity. This supports the call for further research to gain more knowledge about the preferences of food chain stakeholders, to prioritise interventions through the use of specific weighting criteria. For example, when water security is the key concern, closed-loop fertigation should be selected to mitigate water resource depletion. Instead, cascade cropping should be chosen when the economic development is the priority, as it offers the greatest returns to the farmer.

Two critical aspects emerge across all scenarios, which suggest general recommendations: (i) the greenhouse infrastructure is the major source of environmental impacts across all scenarios, then action should be taken to sustain the use of greenhouse materials with extended useful life; (ii) farmer price for the produced vegetable is a sensitive parameter that can markedly affect the decision towards the adoption of technological innovation; supply contracts that acknowledge the sustainability attributes of greenhouse vegetable production might reduce the risk of price fluctuation.

The application of eco-efficiency analysis can support the identification of viable pathways to achieve greater sustainability in greenhouse cropping systems and contribute to continuous method advancement. However, absolute values should be considered with caution, due to study limitations. Some limitations are study-specific, such as the geographical and temporal boundaries of data, the assumptions underlying scenario building, and the use of simplified models for the calculation of emissions from fertilisers and pesticides. Others are more general, such as the existence of data gaps and the lack of context-specificity of background databases and characterisation factors, the limited comparability of life cycle impact assessment results calculated using different methods.

However, the presented research is affected by three main limitations, i.e. (i) the reliance on a representative farm; (ii) the reduced comparability with published articles; (iii) the selection of the background databases for building the cradle-to-gate life cycle inventory and of life cycle impact assessment method. To some extent, the comparative nature of this study reduces the importance of those limitations, as research findings focus on the potential improvements that can be achieved compared to a baseline situation. Nevertheless, absolute impact assessment results should be considered with caution, when compared to the published literature. Those limitations highlight the need for further research to provide more ex-post assessments of reuse technology adoption in the real-world. More assessments are needed covering farms with different characteristics. Additionally, published LCA research should follow agreed and harmonised rules to facilitate the generation of external validity from case studies and then the delivery of more general recommendations.

Overall, this study suggests some directions for further research: (i) to extend case-based assessments to different geographical and social contexts, including those not currently covered by background databases; (ii) to calibrate scenario-based life cycle inventory models using real-world data; (iii) to involve multidisciplinary stakeholders through participatory methods to identify socially and financially acceptable interventions for assessment; (iv) to develop specific weighting frameworks to deal with trade-offs among environmental impacts to support the prioritisation of interventions based on local conditions and stakeholder needs; (v) to develop win-win supply contracts for farmers and retailers.

Annex 1 - Theory

This study develops an eco-efficiency analysis by integrating LCA and LCC. LCA and LCC are process-based tools to compile the inventory (quantities, costs) of all the inputs and outputs of crop production and to assess the environmental impacts and natural resource use (LCA) and the economic impacts (LCC), from raw material acquisition to disposal (Finkbeiner et al., 2006; Huguet Ferran et al., 2018; Hunkeler et al., 2008; Swarr et al., 2011). In the LCA, this is done through a stepwise approach with four phases (see Brentrup et al., 2004; Curran, 2013; Pennington et al., 2004; Rebitzer et al., 2004 for more details):

(1) goal and scope definition (including the identification of system boundaries the selection if the functional unit);

(2) life cycle inventory analysis, i.e. the compilation of all the relevant inputs and outputs (including direct and indirect emissions to the environment and consumption of resources);

(3) life cycle impact assessment: the outputs of the inventory are classified according to the effect they have on the environment and assigned to impact categories using characterization factors, representing the potential of specific emissions or resource consumption to contribute to the relative impact category, as follows:

$$IC_i = \sum_j (E_j \vee R_j) \times CF_{i,j}$$

where, IC_i = impact category i; E_j or R_j = emission j or consumption of resource j; $CF_{i,j}$ = characterization factor for E_j or R_j contributing to IC_i. CF are calculated via quantitative models at the midpoint (CF_m) or endpoint (CF_e) level. Endpoints are the attributes or aspects of natural environment's ecosystems, human health, resource availability (areas of protection), identifying the ultimate environmental impacts of concern; midpoints represent the relative contribution of emissions/resource consumption to an endpoint at an earlier point on the cause-effect chain between emissions/resource consumption towards endpoints (JRC, 2012). CF_m and CF_e are calculated, based on fate factors (FF), optional exposure factors (EF), effect factors (EFF) and optional damage factors (DF), as follows (Morelli et al., 2018):

$$CF_m = FF \times EF$$

$$CF_e = CF_m \times EFF \times DF$$

(4) interpretation, to support result understanding and informed decision making in business and policy.

The same steps apply to the LCC, except for impact assessment, as data are already expressed in currency units; instead, attention should be paid to cost grouping and the identification of relevant economic criteria, such as the definition of costs and the selection of the discount rate, when future cash flows are relevant for the assessment, to cite a few (Heijungs et al., 2013; Ristimäki et al., 2013; Swarr et al., 2011).

Annex 2 – Data collection process

The retrieval of farm records (2014-2018) allowed to model the baseline system and to collect most information for the inventory analysis; information about the transportation stage (means of transport and distances) and producer price for tomatoes was asked directly to the farmer via face-to-face interview (cf. Banaeian et al., 2011). The field trial was used to collect additional information for modelling other scenarios, especially CLS, as well for the daily monitoring of the volume of drainage water and its concentration of crop nutrients. Daily figures about drainage water volume and emissions to water were summed through the duration of the crop cycle per year and then averaged over the study period. The analytical methods for emission monitoring are as follows:

- K, Na: flame photometry (Flame Photometer 230 VAC 50, 60 Hz);
- Ca, Mg, Cu, Fe, Zn, Mn: atomic absorption spectrophotometry (Varian Model Spectra-AA240 FS, Australia);
- P, B: colorimetric analysis, i.e. P via molybdate method (Olsen and Sommers, 1982), B via azomethine-H method (Page et al., 1982),
- N: spettrophotomethic analysis, i.e. N-NO_3 via salicylic acid method (Cataldo et al., 1975), N-NH4 via indophenol blue spectrophotometric method (Tartari and Mosello, 1997).

A first round of interviews with experts (farm advisor; a local wholesaler of agricultural supplies, a local agricultural building company, a representative of the local waste company, who were identified by farm advisor) was organised to gather missing information needed to complete system modelling, especially WWTP scenario and sw-r, cp-o, cp-r subscenarios. Missing information (quantities and prices) included adjustments of BAU greenhouse and fertigation stages to develop WWTP and CSC scenarios (including the identification of the second crop and relative nutritional needs); final manufacturing (mixing), physical characteristics, farmer price and end of life management of coir pith slabs; reuse possibilities for substrates; dismantling of infrastructures; location, means of transports and fees for solid waste and wastewater treatment.

The gathered data were then used to build the inventory using the Ecoinvent ® 3.6 database (Wernet et al., 2016) for background processes. For bridging process gaps in the Ecoinvent ® 3.6 database, a literature search was carried out and relevant official and refereed papers used as reference.

Emission factors for the calculation of emissions to air from fertilisers and pesticides from input data were taken from the literature and selected by comparison with similar LCA studies.

A second interview round was carried out face-to-face with different experts. Based on the experience of the research team with farm level studies in horticulture, three farm advisors were selected who operate in the key greenhouse cropping areas of Tuscany, i.e. the provinces of Pistoia, Lucca, and Livorno (NUTS 3 level regions). The experts were asked for feedback on inventory data for both the LCA and the LCC of all scenarios and subscenarios, to verify if they were consistent with the sector, to propose adjustments and to suggest sensitive parameters. The purpose of the identification of sensitive parameters was to extend the usefulness of study findings to the sector. The identified parameters were transport distances, as those observed for the case study were shorter than for most farms, and farmer prices for tomatoes, as they can be affected by relevant fluctuations on a yearly basis.

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Declaration of interests

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

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